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Effects of Grazing and Fire on Soil Microbial Communities and Hydrological Processes in the Northern Great Plains Grassland

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EFFECTS OF GRAZING AND FIRE ON SOIL MICROBIAL COMMUNITIES AND HYDROLOGICAL PROCESSES IN THE NORTHERN GREAT PLAINS GRASSLAND

BY

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THESIS ACCEPTANCE PAGE

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This thesis is approved as a creditable and independent investigation by a candidate for the master's degree and is acceptable for meeting the thesis requirements for this degree. Acceptance of this does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

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ABSTRACT

EFFECTS OF GRAZING AND FIRE ON SOIL MICROBIAL COMMUNITIES AND HYDROLOGICAL PROCESSES IN THE NORTHERN GREAT PLAINS GRASSLAND

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Historic grazing and fire regimes have been altered with the development of the livestock industry in the Northern Great Plains and have resulted in a reduction of diversity across all scales. As alternative land surface disturbances are developed to combat the loss of diversity, their potential to serve as a sustainable land surface disturbance should be evaluated. To determine the ability of an alternative grazing strategy to serve as a sustainable land surface disturbance, the reaction of the soil microbial community and soil hydrological processes should be evaluated. Objectives of this study were to: 1) evaluate the impact of alternative land surface disturbance strategies, high-intensity winter-grazing (WG) and wildfire (WF), compared to a commonly used summer-long continuous grazing (CG), on the soil microbial community, measured by total soil microbial biomass, percent soil microbial functional groups, and soil microbial diversity and 2) determine the impact of the land surface disturbance strategies on soil moisture and temperature, infiltration rates, and erosion processes, which include soil loss, surface runoff, and sediment yield. To determine changes in the soil microbial community, four soil cores were taken from nine exclosures (3 CG, 3 WG, and 3 WF) during the beginning of the growing season (June) and peak growing season (August) for two years (2017 and 2018). Soil moisture and temperature sensors were

installed at three depths (6-, 12, and 24-inch), across replicates of nine land surface disturbance areas (3 CG, 3 WG, and 3WF pastures), and monitored for approximately two years (April 13th, 2017 to December 31st, 2018). Infiltration tests were also performed within each land surface disturbance area every two months from June 2017 to August 2018. Surface runoff, soil loss, and sediment yield were modeled using the Watershed Erosion Prediction Project (WEPP) model. A total of nine scenarios were developed to evaluate the impact of land surface disturbances individually, hillslope individually, and the combined impact of land surface disturbance and hillslope.

Total soil microbial biomass was significantly affected by land surface disturbances but was dependent on vegetation characteristics. Total microbial biomass increased as a result of fire in areas that had a greater percentage of shortgrass species and decreased in areas that had more mid-grass species. Similar to total microbial biomass, the effect of the surface disturbances on microbial functional groups was dependent on vegetation characteristics and differed between land surface disturbance treatments. No differences were found for soil microbial diversity between land surface disturbance treatments. Soil moisture and temperature were affected by land surface disturbance treatments, while infiltration rates were not. Although there were differences between land surface disturbance treatments for soil moisture and temperature, these differences are not likely to have any biological effect with the greatest difference between treatments being 0.9 °C. Results from the WEPP model showed that winter-grazed land surface disturbance scenarios do not drastically increase the amount of surface runoff, soil loss, or sediment yield when compared to continuous summer-long grazing while wildfire has dramatic increases compared to summer-long continuous

grazing. Results of this study show that high-intensity winter-grazing does not cause detrimental impacts on the soil microbial community or soil hydrological processes. However, wildfire may not cause detrimental impacts on the soil microbial community but does increase surface runoff, soil loss, and sediment yield. This suggests that high-intensity winter-grazing could serve as a sustainable land surface disturbance strategy.

CHAPTER 1: INTRODUCTION

The Northern Great Plains (NGP) are one of the largest grassland biomes on earth (Bock et al. 1993) located throughout the states of North Dakota, and South Dakota, with large areas in Montana, Wyoming, and Nebraska, extending up through the southern portions of Canada (Havstad et al. 2009). This large expanse of grassland evolved under fire and grazing regimes. Historically, natural (lightning) and human (Native American) started fires would periodically burn large areas of grassland. Large herds of bison would graze recently burned grassland following new regrowth created by wildfire (Briske 2017). As a result, a natural form of rotational disturbance regimes of grazing and fire occurred. This rotational grazing, in response to fire, created a mosaic of habitats and plant communities which supported diverse populations of plants and wildlife species (Samson et al. 2004).

Changes to Historic Disturbances

Historic disturbances began to shift as European settlement occurred in the NGP and cattle grazing became a dominant land use practice. European settlement altered natural grazing and fire regimes which resulted in changes to ecosystem services and a reduction of heterogeneity across the landscape. Currently, alternative disturbances are needed to restore heterogeneity and improve ecosystem services and diversity.

The Homestead Act of 1862 and the Canada Dominion Land Act of 1872, along with other federal acts that transferred government land to private landowners, resulted in the reduction of grasslands in the NGP due to the conversion of rangeland to agriculture, which includes livestock production (Samson et al. 2004). Since European settlement, livestock grazing has become the most widespread economic land use practice in western North America (Bock et al. 1993). The large increase in cattle production significantly altered historic grazing and fire regimes.

Grazing management, following NGP settlement, focused on increasing livestock productivity by managing for uniform plant communities most favorable to livestock through continuous non-rotational grazing (Fuhlendorf and Engle 2004). Thus, homogenous plant communities consisting of only the plants most palatable and productive to livestock were supported (Fuhlendorf and Engle 2001). This continuous non-rotational grazing practice reduces ecological diversity and the ability of landscapes to provide ecosystem services.

Along with a change in grazing regimes, from natural bison grazing following a fire to continuous non-rotational cattle grazing, the occurrence and intensity of fires were also altered by the development of the livestock industry. Modern livestock practices focus on increasing the predictability of natural systems. To increase this predictability, wildfires were suppressed (Briske 2017). Large scale livestock operations reduced fire-frequency in the NGP through the consumption of fine fuels (Bock et al. 1993), embedding fine fuels into the soil by trampling (Nader et al. 2007), and the creation of fire breaks (Briske 2017). Using fire as a surface disturbance tool is also avoided by livestock producers due to concerns of forage loss, liability, as well as having the necessary labor and equipment to successfully perform prescribed burns on rangelands (Toledo et al. 2014).

In order to combat the loss of diversity, as a result of altered grazing and fire regimes, alternative disturbances are needed in the NGP. Alternative disturbances can allow species with low and high competitive abilities to exist in an ecosystem (Briske et al. 2011). By allowing plants with various competitive abilities to exist, ecosystem services and diversity can be improved.

Ecosystem services are defined as the benefits people obtain from ecosystems and rely directly on diversity across the landscape (Assessment 2005). Fire and grazing are the two major forces driving heterogeneity on rangelands and therefore facilitate grasslands in providing ecosystem services. To preserve the ability for grasslands to continue providing ecosystem services, land management strategies that decrease heterogeneity must be altered.

Rangeland Soil Health

Soil health can be defined as "the capability of a living soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health" (Doran and Zeiss 2000). A specific component of soil health is the soil microbial community, which should be a focus for any land management strategy as it has the capability to influence landscape changes (Briske 2017). The soil microbial community is important because of its influence on many ecosystem processes and conditions (Atlas 1998). Studies focusing on heterogeneity have often focused on plant communities and vertebrates but not soil microbial community diversity (Baskin 1994). Attention should be paid to microbes as they can determine soil carbon accumulation, pH, and rates of C and N transfer between plant roots and soil organism's in addition to influencing soil hydrologic processes (Briske 2017). The soil microbial community is influenced by grazing and fire disturbances as well as by changes in the plant community (Zak et al. 2003). Grazing can influence the soil microbial community through the removal of aboveground vegetation, animal trampling, and manure deposition (Yang et al. 2013). Areas that are moderately grazed have been shown to be dominated by more bacterial communities whereas areas that are intensively grazed are dominated by fungal communities (Bardgett et al. 2001). By changing plant community composition, grazing influences the soil microbial community and diversity (Belsky 1992). Due to the influence of grazing on the soil microbial community, studies on adaptive grazing strategies should include their effect on the soil microbial community.

The reaction of the soil microbial community in response to fire has had mixed results as the intensity and frequency of fires determines the severity of impact on the soil microbial community (Staddon et al. 1996). As fire intensity increases the soil microbial community is reduced (D'Ascoli et al. 2005). However, this reduction of the soil microbial community is temporary, and the soil microbial community functional diversity recovers quickly after fire regardless of the magnitude of initial reduction (Doerr and Cerdà 2005). Although many direct links have been made to fire influencing soil microbial communities, others have demonstrated little to no effect due to the many abiotic and biotic factors that influence the soil microbial community (Lindeburgh 1990, Staddon et al. 1996).

Although the plant community determines the soil microbial community, the soil microbial community plays an important role in maintaining plant community diversity (Bever et al. 1997). Soil microbial communities maintain plant community diversity by influencing the ability of plants to take up nutrients. Soil microbial communities occur largely in the root-soil interface and can mediate mineral-microbe-root chemical exchanges (Balogh-Brunstad et al. 2008). This relationship creates soil microbial environments that are resilient and capable of maintaining soil microbial and terrestrial ecosystems (Schimel et al. 2007).

Rangeland Soil Water and Erosion Regulation

One of the most fundamental services provided by ecosystems is supplying the quantity and quality of water needed to provide ecosystem services (Falkenmark et al. 2004, Brauman et al. 2007). The ability of the soil community on rangelands to provide ecosystem services is highly dependent on the availability of water (Campbell and Allen-Diaz 1997). Precipitation that falls on rangelands supports the people, livestock, and wildlife in the surrounding area (Le Maitre et al. 2007). Precipitation can either infiltrate, evapotranspire, or move as overland flow. The path of precipitation is determined based on vegetation structure, which can be altered through grazing and wildfire disturbances (Briske 2017).

Rangeland hydrologic processes are capable of changing the structure of soil microbial communities and are also impacted by the soil microbial community. One component of hydrologic processes that alters soil microbial communities is soil moisture (Reichel et al. 2014). During periods of drought, the soil microbial community may shift towards a community composed of microbes more resilient to drought (Bérard et al. 2012). Infiltration rates are a component of rangeland hydrologic processes that can be influenced by the soil microbial community. The soil microbial community can impact infiltration rates as a result of biological processes, such as the formation of hydrophobic compounds produced by fungi (Doerr and Cerdà 2005, Schnabel et al. 2013), which has the potential to substantially decrease infiltration (Doerr and Cerdà 2005).

Vegetation structure also affects rangeland hydrologic processes by slowing overland flow, which allows water more time to infiltrate the soil (Briske 2017).
Any water that does not infiltrate the soil becomes surface runoff. Due to the large size of rangeland ecosystems, the 5 percent of precipitation that becomes surface runoff is substantial (Wilcox et al. 2003a). The spatial heterogeneity of rangelands has large impacts on the amount of runoff that occurs. If disturbance creates areas of bare ground or patchy vegetation, then the amount of runoff will increase proportionally to the amount of vegetation reduced (Wilcox et al. 2003b). Important interactions between vegetation patches and runoff occur at the hillslope level since this is where vegetation patches can catch and slow surface runoff (Briske 2017). If disturbances occur that alter vegetation patch structure, such as overgrazing or fire, the hydrological process may be disrupted.

Disturbance Effects on Rangeland Hydrology

Disturbance on rangelands creates the conditions that increase erosion and runoff (Belnap et al. 2014). In areas that have been undisturbed, protective layers, in the form of biocrust, vegetation cover, and litter form and protect the soil from erosion (Briske 2017). When disturbances occur in areas that have these protective layers their resilience to disturbance decreases and they become susceptible to further disturbances (Briske 2017). For example, fires can create extremely hot soil surface temperatures that volatize compounds, creating a water repellant layer at the soil surface (Ice et al. 2004). The effect of this water-repellant layer decreases water infiltration and increases the potential for erosion and runoff. Fires also remove vegetation structure which can create concentrated flow paths which accelerate erosion (Doerr et al. 2000, Schnabel et al. 2013). Grazing is another disturbance that impacts rangeland hydrology. Intense grazing decreases infiltration rates and soil water content through soil compaction (Trimble and Mendel 1995); whereas light to moderate grazing has little effect on soil infiltration on rangelands (Hiernaux et al. 1999, Ludwig et al. 1999). Increased compaction from intense winter grazing (Derner and Schuman 2007) increases runoff and overland flow (Wilcox 2002). Intense grazing also disturbs soil aggregates that are responsible for maintaining soil porosity (Briske et al. 2011) and is more associated with stocking rate rather than the duration or season (Briske 2017). As a result of the potential disruption disturbance can cause on rangeland hydrology, the impacts of any new land management strategy, on rangeland hydrology, should be investigated.

Research Overview

The purpose of this study is to determine soil microbial community structure and soil hydrology response to summer-long continuous grazing, high-intensity winter-grazing, and a wildfire. An alternative grazing strategy was developed to attempt to mimic the effects of fire (such as the reduction of litter) with high-intensity winter-grazing. A wildfire occurred at the Cottonwood Field Station allowing us the opportunity to incorporate fire into this study and evaluate its effects on soil microbial communities and soil hydrology.

Our first objective of this study was to examine soil microbial community functional structure and group diversity response to land surface disturbance treatments.

This was conducted through phospholipid fatty acid analysis (PLFA). Soil microbial functional groups were identified and differences between treatments were analyzed. We hypothesize that the soil microbial community diversity will decrease initially in the high-intensity winter-grazed and wildfire treatments but increase to a greater diversity two years post-disturbance when compared to the summer-long continuous grazing. This is expected due to high-intensity winter-grazing and wildfire potentially creating diverse plant communities which will, in turn, create diverse soil microbial communities (Perkins and Nowak 2013).

Our second objective is to determine if our surface disturbance treatments affect soil and hillslope hydrologic processes. Soil hydrology will be measured by soil temperature, moisture, and infiltration rates. Hillslope runoff and erosion will be modeled using windows interface erosion prediction software (WEPP). We hypothesize soil temperature to be higher in intense winter grazing and wildfire burned treatments when compared to summer-long continuous land surface disturbance treatments. The soil will become warmer in these patches as a result of solar heating (Neary et al. 1999). Due to this hypothesized soil warming, we expect soil moisture will be significantly lower in alternative land surface disturbance treatments than summer-long continuous grazing.

We hypothesize high-intensity winter-grazing and wildfire land surface disturbance treatments will decrease infiltration rates and lead to higher amounts of runoff and erosion compared to summer-long continuous grazing. Infiltration rates in high-intensity winter-grazed and wildfire burned areas are hypothesized to recover following the first growing season. High-intensity winter-grazed and wildfire land surface disturbance treatments were studied following each disturbance to determine the capability of infiltration rates to recover following each disturbance.

Finally, we hypothesize that high-intensity winter grazing hillslopes will result in slightly more runoff and erosion than the summer-long grazing. Runoff and erosion for high-intensity winter-grazed surface disturbance treatments are not hypothesized to be significantly higher than the summer-long continuous grazing due to vegetation structure and litter still being maintained. We hypothesize hill slopes that experience wildfire will have significantly higher amounts of runoff and erosion due to the loss of all vegetation structure and litter cover. This hypothesis will be tested with the Watershed Erosion Prediction Project (WEPP) computer interface erosion prediction software.

CHAPTER 2: EFFECTS OF GRAZING AND FIRE ON SOIL MICROBIAL COMMUNITIES IN THE NORTHERN GREAT PLAINS GRASSLAND

Abstract

Grazing and fire in the Northern Great Plains grasslands can influence the soil microbial community. The soil microbial community can form important relationships with rangeland plants that improve ecosystem services. As grazing and fire regimes have been altered, rangelands have experienced a loss of diversity across all scales. Recently, alternative land surface disturbance strategies are being used to diversity. To combat this loss of diversity alternative land surface disturbance treatments are being implemented. This study assessed the impact of an alternative grazing land surface disturbance treatment, high-intensity winter-grazing (winter 2016 – 2017, WG), and a wildfire (October 2016, WF), compared to a widely used conventional summer-long continuous grazing (non-burned and non-intensely grazed pastures; CG), on changes in total soil microbial biomass, soil microbial functional groups, and soil microbial diversity. The soil microbial community was evaluated at beginning of the growing season (June) and peak growing season (August) for two years (2017 and 2018) following the treatments. Prior to the treatments, the pastures had summer-long continuous grazing. Our results indicate that the soil microbial community is fairly resistant to land surface disturbance treatments (high-intensity winter-grazing and wildfire) although the soil microbial community response was different in areas with different vegetation. Total microbial biomass increased as a result of wildfire in areas that had a greater percentage of shortgrass species and decreased in areas that had more mid-grass species. Similar to total microbial

biomass, the effect of the land surface disturbance treatments on microbial functional groups were dependent on vegetation. Neither high-intensity winter grazing nor wildfire affected the diversity of soil microbial functional groups. Overall, neither wildfire nor winter-grazing caused significant impacts on the soil microbial community.

Introduction

The soil microbial community plays an important role in maintaining ecosystem services. Ecosystem services are defined as the benefits that people obtain from ecosystems, whether this is production from agriculture which includes livestock production, clean water and air, recreation, or aesthetics (Assessment 2005). Studies focusing on ecosystem services have often focused on the plant community (Atlas 1998). However, the soil microbial community should also receive attention as it has the capability to influence many ecosystem services. For example, soil microbial diversity promotes plant growth and plant species diversity (Bever et al. 1997). The soil microbial community also drives cycles in nutrients and carbon (Kowalchuk and Stephen 2001, Van Der Heijden et al. 2008) and suppresses disease (Mendes et al. 2011). Because of the importance of the soil microbial community to ecosystem services, understanding how different common land surface disturbance practices (such as grazing and fire) affect the soil microbial community is important.

Results of studies on the reaction of the soil microbial community to grazing are variable (Milchunas and Lauenroth 1993, Derner et al. 2006). Grazing can influence the soil microbial community by controlling the quality and quantity of resources that enter the soil through the removal of aboveground vegetation, animal trampling, and nutrient deposition (Ayres et al. 2007, Yang et al. 2013). Grazing also alters the aboveground and below ground location of plants along with root physiology and biomass, which have a large impact on the soil microbial community (Veen et al. 2014). Although grazing has the capability to influence these soil properties, the degree to which they are altered varies based on the intensity of grazing. Depending upon grazing regimes and intensity, soil microbial biomass can increase, decrease, or be unaffected (Bardgett et al. 2003, Mills and Sina Adl 2006, Wang et al. 2006).

Fire can create short, medium, or long-term changes to the soil microbial community (Neary et al. 1999, González-Pérez et al. 2004). The effect of fire depends on fire frequency and intensity (Staddon et al. 1996). Intense wildfires, that are hotter than prescribed fire, can cause detrimental impacts to the soil microbial community, such as reducing total soil microbial biomass. Less intense prescribed fires may cause lower detrimental impact on the soil microbial community (Dooley and Treseder 2012). A reduction of total soil microbial biomass could negatively affect the resistance of soils to disturbance events. Although there may be different initial effects of fire on total soil microbial biomass immediately following a fire, the functional diversity of the soil microbial community recovers quickly (D'Ascoli et al. 2005). The reaction of the soil microbial community, however, relies on environmental conditions. In periods of drought, microbes can enter a dormant state and become more resistant to the effects of fire (Mataix-Solera et al. 2009). Due to different effects fire can have on the soil microbial community, depending on the frequency and intensity of the fire, along with the variable weather conditions that affect the reaction of the soil microbial community, the effect of fire on the soil microbial community should be evaluated.

In this study, we evaluated three land surface disturbance treatments (summer-long continuous grazing, high-intensity winter-grazing, and wildfire) on the soil microbial community in regard to total soil microbial biomass, soil microbial community composition, and soil microbial diversity. Our specific objective of this research was to determine if the soil microbial community was negatively affected by two alternative land surface disturbance treatments (high-intensity winter-grazing and wildfire) compared to the commonly used summer-long continuous grazing (control). We hypothesize that the high-intensity winter-grazing and wildfire will cause a decline in overall soil microbial biomass and result in less microbial diversity immediately following the land surface disturbance treatments. Further, we hypothesize total soil microbial biomass will recover the growing season after the initial land surface disturbance treatment and will improve soil microbial diversity. Overall, we expect that alternative surface disturbance treatments will not cause significant detrimental impacts on the soil microbial community.

Methods

Study Area:

This research occurred at the Cottonwood Field Station in Cottonwood, South Dakota. Cottonwood is located approximately 120 km east of Rapid City, S.D and is in the Northern Great Plains mixed-grass prairie. The topography of the study area is gently sloping with long, rolling hills and relatively flat-topped ridges. The major vegetation includes western wheatgrass (*Pascopyrum smithii*), green needlegrass (N*assella viridula*), buffalograss (*Bouteloua dactyloides*), and blue grama (*Bouteloua gracilis*). The study site climate is characterized as continental and semi-arid with hot summers and cold winters. The hottest month occurs in July with an average high of 29 °C and an average low of 15 °C. The coldest month is in January with an average high of -6°C and an average low of -18 °C. The thirty-year average annual precipitation for the area is 426 mm with about 58% of the precipitation occurring May through August (Mesonet 2019).

Land Surface Disturbance *Treatments:*

The three land surface disturbance treatments evaluated include summer-long continuous grazing (CG-control), high-intensity winter-grazing (WG), and wildfire (WF). Treatments were implemented with a randomized complete block design in three separate pastures. The three pastures ranged in size from 60 ha (130 ac) to 73 ha (180 ac). The vegetation in two pastures consisted of approximately 8% shortgrass species while the other pasture consisted of approximately 24% shortgrass species. Therefore, rather than blocking by pasture, the two pastures with similar vegetation communities were blocked together. The two pastures with approximately 8% shortgrass species were defined as the mixed-grass block and the single pasture with approximately 24% shortgrass species was defined as the shortgrass block. Prior to wildfire and high-intensity winter grazing land surface disturbance treatments being applied, the entire study area was grazed with approximately 1.04 animal unit months per hectare (AUM/ha). The treatments were applied to approximately 1/3 of the area in each pasture. First, a third of each pasture was not grazed again after the initial summer-long continuous grazing and was used for the control. Second, a high-intensity wildfire occurred in October of 2016, prior to the implementation of high-intensity winter-grazing surface disturbance treatments, and burned through approximately a 1/3 of the area in all 3 pastures. Each pasture had approximately equal areas burned by the wildfire. The area burned by the wildfire was not grazed during this study. The final third of each pasture was winter grazed at an extremely high intensity between January 1st, and February 28th, 2017. Winter-grazed areas were stocked with 120 cows on approximately 20 ha during the winter months for an approximate 2.4 AUM/ha (January and February). Cattle were removed from the winter grazed areas when standing vegetation had been reduced to an approximate height of 8-10 cm, which occurred in approximately 30 days. Sampling exclosures, approximately 5 m^2, were placed in each land surface disturbance treatment area where all three treatments converged (Fig 1). Exclosures prevented further grazing from livestock. Samples were collected adjacent to other treatments to reduce environmental variability.

Microbial Community Sampling (PLFA)

Four randomly placed soil cores (2.5 cm diameter, 10 cm deep) were taken from each treatment area (Fig 1) during mid-June and late-July for two consecutive years (2017 and 2018), for a total 48 soil samples per treatment, 144 samples total. The collection times coincided with beginning and peak growing seasons. Litter was removed before soil samples were collected. No standing vegetation was removed. Samples were placed in sterile sampling bags and immediately frozen. Samples were then sent overnight to the Microbial ID Laboratory, Inc (Smithwick et al. 2005, Perkins and Nowak 2013).

The soil microbial community was then determined chemotaxonomically with phospholipid fatty acid analysis (PLFA) using a hybrid PLFA and fatty-acid methyl ester (FAME) technique (Bligh and Dyer 1959, Smithwick et al. 2005) and gas chromatography. PLFA concentrations were determined by comparing sample peaks to a 13:0 fatty acid methyl ester (FAME) standard (Perkins and Nowak 2013). Chemotaxonomic grouping of PLFAs followed Zelles (1999) and Mitchell et al. (2010), (Perkins and Nowak 2013): 13:1 w5c, 14:1 w8c, 14:1w5c, 15:1 w9c, 15:1 w6c, 16:1 w9c, 16:1 w7c, 17:1 w8c, 17:0 cyclo w7c, 16:0 2OH, 18:1 w7c, 18:1w6c, 18:1 w5c, 19:1 w8c, 19:0 cyclo w7c, 20:1 w9c, 20:1 w8c, 20:1 w6c, 20:1 w4c, 21:1 w8c, 21:1 w3c,22:1 w9c, 22:1 w8c, 22:1 w3c, and 24:1 w9c were considered indicative of gram negative bacteria;13:0 iso, 14:0 iso, 15:1 iso w6c, 15:1 anteiso w9c, 15:0 iso, 15:0 anteiso, 16:0 iso, 16:0 anteiso, 17:1 iso w9c, 17:0 iso, 17:0 anteiso, 18:0 iso 17:1 anteiso w9c, 17:1 anteiso w7c, 19:0 iso, 20:0 iso, 22:0 iso were indicative of gram- positive bacteria; 16:0 10-methyl, 17:1 w7c 10-methyl, 18:1 w7c 10-methyl, 19:1 w7c 10-methyl, 20:0 10-methyl were indicative of actinomycetes; and 18:2 w6c was indicative of fungi. Forty-nine different PLFA biomarkers were detected and identified. The total number of carbon atoms are used to identify fatty acids. The degree of unsaturation is indicated by a number separated from the chain length number by a colon. The degree of unsaturation is followed by either *x,* to indicate the position of the double bond closest to the carboxyl end, or *w*, to indicate the position of the double bond is closest to the aliphatic end (Zelles 1999). Prefixes, *a, i, cy,* and *d* are used to identify the branching of fatty acids and refer to *antesio, iso, cyclopropyl,* and *dicarboxylic* respectively. If branching type is unknown the prefix *br* is used.

PLFA analysis is not capable of identifying microorganisms at a species level (Frostegård et al. 2011, Perkins and Nowak 2013) but can provide a snapshot of the microbial community and identify changes in microbial functional group composition. Therefore, the classification of PLFA biomarkers should be considered an indicator of specific microbial functional groups and changes in PLFA are indicative of changes to the entire soil microbial community (Perkins and Nowak 2013).

Microbial Diversity

The diversity of the microbial community in response to land surface disturbance treatments was compared using the Shannon Wiener Diversity Index (H). H was calculated by:

$$H = -\sum[(p_i) * ln(p_i)]$$

Where $\sum =$ summation and $p_i =$ the proportion of individual biomarkers found in functional group *i*. Shannon Wiener diversity was calculated for each soil sample collected over the entire study period.

Statistical Analysis

For analysis purposes, the pastures were split into groups based on vegetation characteristics. To determine which factors influence soil microbial biomass, functional group percent, and microbial diversity, a full model analysis was conducted. Response variables in the full model include total soil microbial biomass, percent fungi, percent actinomycetes, percent gram-positive bacteria, percent gram-negative bacteria, and Shannon Wiener diversity index. Explanatory variables of the full model include treatment, block, season, and year. The interaction effects of the full model include treatment * block, treatment * season, and treatment * year. From these explanatory variables, block had the greatest overall effect on the soil microbial community. Further analysis of total soil microbial biomass, each soil microbial functional group, and soil microbial diversity was split by block and included the explanatory variables: treatment, season, year, treatment * season, treatment * year, and treatment * season * year.

Normality tests (Shapiro/Wilk) were performed on residuals of total soil microbial biomass, each soil microbial functional, group and Shannon-Weiner diversity indices within each block. The *p*-value for Shannon-Weiner diversity indices and microbial functional groups were above $\alpha = 0.05$ showing a normal distribution of the residuals, permitting us to conduct Analysis of Variance (ANOVA) models for individual blocks. We performed ANOVA with treatment (CG, WG, WF), season (spring and summer) and year (2017 and 2018) as main effects. A full factorial with treatment was included in the analysis. Interaction effects included treatment * season, treatment * year, and treatment * season * year. Where significant differences between model effects occurred a post-hoc Tukey Honest Significant Difference (HSD), or Least Square (LS) means student's t-test was performed to determine which model effects differed from one another.

Results

Total Microbial Biomass

Land surface disturbance treatments and season had significant effects on total soil microbial biomass in both the mixed-grass (Table 1) and the shortgrass blocks (Table 2). However, in the mixed-grass block, the interactions of treatment * season and treatment * year were significant. In the mixed-grass block treatment * season and treatment * year were significant. Winter grazed land surface disturbance treatments, combined, begin with higher total microbial biomass in the spring and in 2017. During the summer and in 2018, control land surface disturbance treatments had higher total microbial biomass than winter grazed land surface disturbance treatments (Fig. 2a). In the mixed-grass block, total soil microbial biomass was significantly higher in 2018 than 2017. Within both blocks, total microbial biomass increased from spring to summer (Fig. 2a, 3a). In the shortgrass block, total microbial biomass was highest under wildfire land surface disturbance treatments and lowest under the winter grazed land surface disturbance treatment (Fig. 3a)

Soil Functional Group Composition

Season and year significantly influenced the percent of gram-negative bacteria in the mixed-grass (Table 1) and the shortgrass block (Table 2). The percent of gram-negative bacteria increased from spring to summer in the mixed-grass (Fig. 1b) and shortgrass block (Fig. 2b). The percent of gram-negative bacteria also increased from 2017 to 2018 in the mixed-grass (Fig. 2b) and shortgrass block (Fig. 3b).

Year significantly affected the percent of gram-positive bacteria in the mixed-grass (Table 1) and shortgrass block (Table 2) with average decreases in both blocks from 2017 to 2018 (Fig 2c, Fig. 3c). In addition to year, treatment and season have significant effects on the percentage of gram-positive bacteria in the mixed-grass block. Combined, control surface disturbance treatments had the highest percent of gram-negative bacteria while wildfire had the lowest percent of gram-positive bacteria. In the mixed-grass block, gram-positive bacteria, combined, decreased from spring to summer.

Season and year both significantly affected percent fungi in the mixed-grass (Table 1) and shortgrass block (Table 2). Percent fungi, combined, increased from spring to summer and 2017 to 2018 in both blocks (Fig 2d, 3d). Additionally, in the mixed-grass block, treatment * season and treatment * year were significant. Combined, winter grazed surface disturbance treatments had the lowest percent fungi in the spring but had higher percent fungi in the summer than wildfire surface disturbance treatments (Fig. 2d). The same occurs in 2017 when winter grazed surface disturbance treatments, combined, have the lowest percent fungi but have higher percent fungi than wildfire surface disturbance treatments in 2018.

Season and year significantly affect the percentage of actinomycetes in the mixed-grass (Table 1) and shortgrass block (Table 2). Together, the percentage of actinomycetes decreased from the spring to summer in the mixed-grass (Fig. 2e) and shortgrass block (Fig. 3e). The percent of actinomycetes also decreased from 2017 to 2018 in both blocks (Fig 2e, 3e).

Soil Microbial Community Diversity

No model effects significantly affected the Shannon Winer diversity index in the mixed-grass block (Table 1). However, the three-way interaction, treatment * season * year, significantly affected the Shannon Winer diversity index in the shortgrass block (Table 2). This three-way interaction was significant due to the variability between seasons and years across the entire study period. Although there was variability in the diversity of the surface disturbance treatments across the study the variability was minimal and is not considered to have any biological significance on the soil microbial community diversity.

Discussion

The results of this experiment suggest that the soil microbial community is fairly resistant to land surface disturbance treatments (high-intensity winter-grazing and wildfire) compared to commonly used summer-long continuous grazing. Interestingly, the effects of the alternative surface disturbance treatments on soil microbial biomass and functional group composition were different in our mixed-grass block and in our shortgrass block. For example, soil microbial biomass was decreased by wildfire in the mixed-grass block and increased by wildfire in the shortgrass block. We hypothesized that the surface disturbance treatments would increase soil microbial diversity which was not observed in our data; conversely, no negative effects of the surface disturbance treatments on soil microbial diversity were found either. This was surprising as grazing and fire have been shown to affect the soil microbial community in other settings (Staddon et al. 1996, Guerrero et al. 2005, Derner et al. 2006). Our results suggest that alternative land surface disturbance treatments of winter-grazing and wildfire will not negatively impact soil microbial diversity, but the specific effects on soil microbial biomass and soil functional group composition are context-dependent and may be different in different vegetation communities.

In our study, land surface disturbance treatments changed soil microbial group composition differently depending on the plant community. Two reasons for the difference in the effect of wildfire on the soil microbial community in the shortgrass block versus the mixed-grass block can be hypothesized. First, wildfire increases the temperature of the soil surface and can reduce soil microbial biomass by making the soil microbes become dormant as a result of increased soil temperatures (Guerrero et al. 2005). In the shortgrass block, this reduction of soil microbial biomass was not observed following wildfire when compared to summer-long continuous grazing and winter grazed treatments. Therefore, it is possible that the wildfire temperatures were not as hot in that pasture. However, because this was a wildfire, we were not able to measure fire temperature. Second, plant communities with a higher percentage of shortgrass species are more resilient to wildfire disturbances because shortgrass species persist in higher quantities immediately following a wildfire than mid-grass species (Gibson and Hulbert 1987). Wildfire can also increase plant germination (Wright et al. 1982). Therefore, the shortgrass block may have had higher plant growth immediately after the fire which stimulated soil microbial biomass production.

Other factors, such as season and year also significantly affected the soil microbial community and total soil microbial biomass. Increasing soil microbial biomass from spring to summer was expected and has been observed elsewhere (Bossio et al. 1998). Interannual variability can also create differences seen in the microbial community. At our study location, 32.8 cm (12.9 in) of precipitation occurred in 2017 while 2018 had 44 cm (17.3 in) of precipitation. Interannual variation in precipitation can cause changes in the soil microbial community and lower precipitation can reduce total soil microbial biomass (Preece et al. 2019). However, lower total soil microbial biomass was only observed in the drier year (2017) in our mixed-grass block and not in the shortgrass block. Perhaps, no differences in total soil microbial biomass between years were found in the shortgrass block, due to plant communities that have a higher amount of shortgrass species having a higher resilience to weather events (Derner et al. 2008).

Our results suggest that these soils have low diversity in the biomarkers for each microbial functional group and this diversity was not impacted by land surface disturbance treatment. The lack of change in diversity could be due to the resilience (the ability to recover quickly) of the soil microbial community following disturbance (Doerr and Cerdà 2005). However, this could also be due to the methods used to evaluate the soil microbial community. Our methods (PLFA) only allow the examination of functional groups, not microbial species. Perhaps other methods (genetic) would detect changes in species-level diversity due to surface disturbance treatments.

The lack of a consistent effect of surface disturbance treatments on the soil microbial community could be a result of our surface disturbance treatments only occurring once and the ability of the soil microbial community to recover quickly after disturbance events (Doerr and Cerdà 2005, Briske 2017). The quick recovery of the soil microbial community may mask the initial effects of the surface disturbance treatments. Samples collected immediately before and after each disturbance event may be needed to detect the immediate effects of winter grazing and wildfire. Due to the land surface disturbance treatments location impacting the effectiveness of the surface disturbance treatment, the location of the land management treatment should be considered.

Overall, we can conclude that winter-grazed or wildfire land surface disturbance treatments did not cause detrimental impacts to the soil microbial measured by total soil microbial biomass, functional group composition, and microbial functional group diversity. In other words, our results suggest that the soil microbial community is fairly resistant to intense winter-grazing and wildfire. However, the soil microbial community in areas with less shortgrass vegetation did respond to grazing and fire differently than the soil microbial community in areas with greater amounts of shortgrass vegetation. Therefore, we suggest investigating site characteristics interaction with surface disturbance treatments to determine the usefulness of alternative land surface disturbance treatments.

Table 1:

	Mixed-Grass Block											
	Total Microbial Biomass		% Gram-Negative		% Gram-Positive		% Fungi		% Actinomycetes		Shannon-Wiener Diversity	
	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P
Treatment	$10.29_{(2,85)}$	**<0.001**	$0.64_{(2,85)}$	0.53	$7.48_{(2,85)}$	**0.001**	$1.72_{(2,85)}$	0.19	$0.13_{(2,85)}$	0.88	$1.19_{(2,85)}$	0.31
Season	$17.96_{(1,8)}$	**<0.001**	$24.57_{(1,85)}$	**<0.001**	$8.14_{(1,85)}$	**0.005**	$21.47_{(1,85)}$	**<0.001**	$6.02_{(1,85)}$	**0.02**	$0.16_{(1,85)}$	0.69
Year	$12.89_{(1,85)}$	**0.006**	$33.83_{(1,85)}$	**<0.001**	$34.13_{(1,85)}$	**<0.001**	$22.65_{(1,85)}$	**<0.001**	$20.69_{(1,85)}$	**<0.001**	$0.27_{(1,85)}$	0.61
Treatment * Season	$3.77_{(2,85)}$	**0.027**	$0.39_{(2,85)}$	0.68	$0.47_{(2,85)}$	0.6242	$1.08_{(2,85)}$	0.35	$0.42_{(2,85)}$	0.66	$0.04_{(2,85)}$	0.96
Treatment * Year	$4.76_{(2,85)}$	**0.0109**	$0.33_{(2,85)}$	0.72	$0.69_{(2,85)}$	0.51	$0.40_{(2,85)}$	0.67	$0.67_{(2,85)}$	0.67	$0.39_{(2,85)}$	0.68
Treatment * Season*Year	$2.51_{(2,85)}$	0.09	$0.04_{(2,85)}$	0.96	$0.40_{(2,85)}$	0.67	$1.56_{(2,85)}$	0.22	$0.93_{(2,85)}$	0.4	$1.78_{(2,85)}$	0.17

F-value and p-value of treatment, season, year, and all factorial combinations with treatment, on total microbial biomass, percent gram-negative bacteria, percent gram-positive bacteria, percent fungi, percent actinomycetes, and the Shannon Wiener diversity index in the mixed-grass block. Bold values are significant to $P<0.05$.

Table 2:

	Short-Grass Block											
	Total Microbial Biomass		% Gram-Negative		% Gram-Positive		% Fungi		% Actinomycetes		Shannon Wiener Diversity	
	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P	$F_{(DF)}$	P
Treatment	$7.61_{(2,37)}$	**0.0017**	$0.006_{(2,37)}$	0.99	$2.81_{(2,37)}$	0.07	$10.21_{(2,37)}$	**0.003**	$0.20_{(2,37)}$	0.82	$0.58_{(2,37)}$	0.58
Season	$18.25_{(1,37)}$	**0.001**	$8.12_{(1,37)}$	**0.007**	$2.34_{(1,37)}$	0.13	$39.91_{(1,37)}$	**<0.001**	$16.49_{(1,37)}$	**0.002**	$1.45_{(1,37)}$	0.34
Year	$0.0004_{(1,37)}$	0.98	$14.08_{(1,37)}$	**<0.001**	$10.39_{(1,37)}$	**0.003**	$58.93_{(1,37)}$	**<0.001**	$28.72_{(1,37)}$	**<0.001**	$0.54_{(1,37)}$	0.47
Treatment * Season	$0.52_{(2,37)}$	0.6	$0.07_{(2,37)}$	0.93	$0.19_{(2,37)}$	0.83	$8.26_{(2,37)}$	**0.001**	$0.80_{(2,37)}$	0.46	$0.62_{(2,37)}$	0.54
Treatment * Year	$0.62_{(2,37)}$	0.54	$0.71_{(2,37)}$	0.5	$1.30_{(2,37)}$	0.37	$2.13_{(2,37)}$	**0.13**	$0.07_{(2,37)}$	0.94	$0.19_{(2,37)}$	0.83
Treatment * Season*Year	$1.55_{(2,37)}$	0.22	$0.91_{(2,37)}$	0.41	$1.30_{(2,37)}$	0.28	$1.99_{(2,37)}$	0.15	$0.16_{(2,37)}$	0.85	$12.27_{(2,37)}$	**<0.001**

F-value and p-value of treatment, season, year, and all factorial combinations with treatment, on total microbial biomass, percent gram-negative bacteria, percent gram-positive bacteria, percent fungi, percent actinomycetes, and the Shannon Wiener diversity index in the shortgrass block. Bold values are significant to P<0.05.

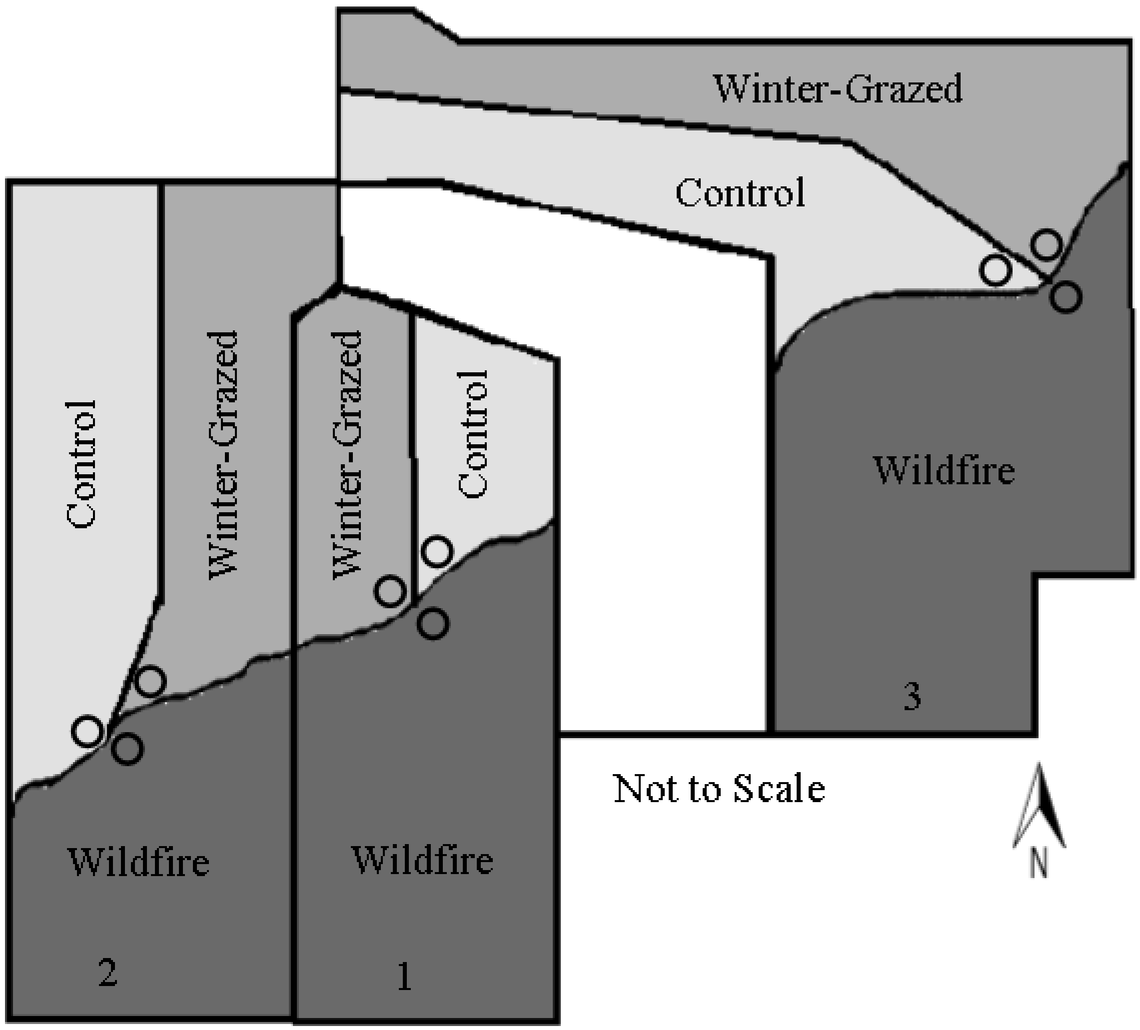


Figure 1:

Map showing field data collection sites at Cottonwood Field Station in Cottonwood, South Dakota to evaluate summer-long continuous gazing (Control), winter grazing, and wildfire disturbances on soil microbial communities. Circles indicate the locations of a series soil test in each pasture.

Mixed-Grass Block

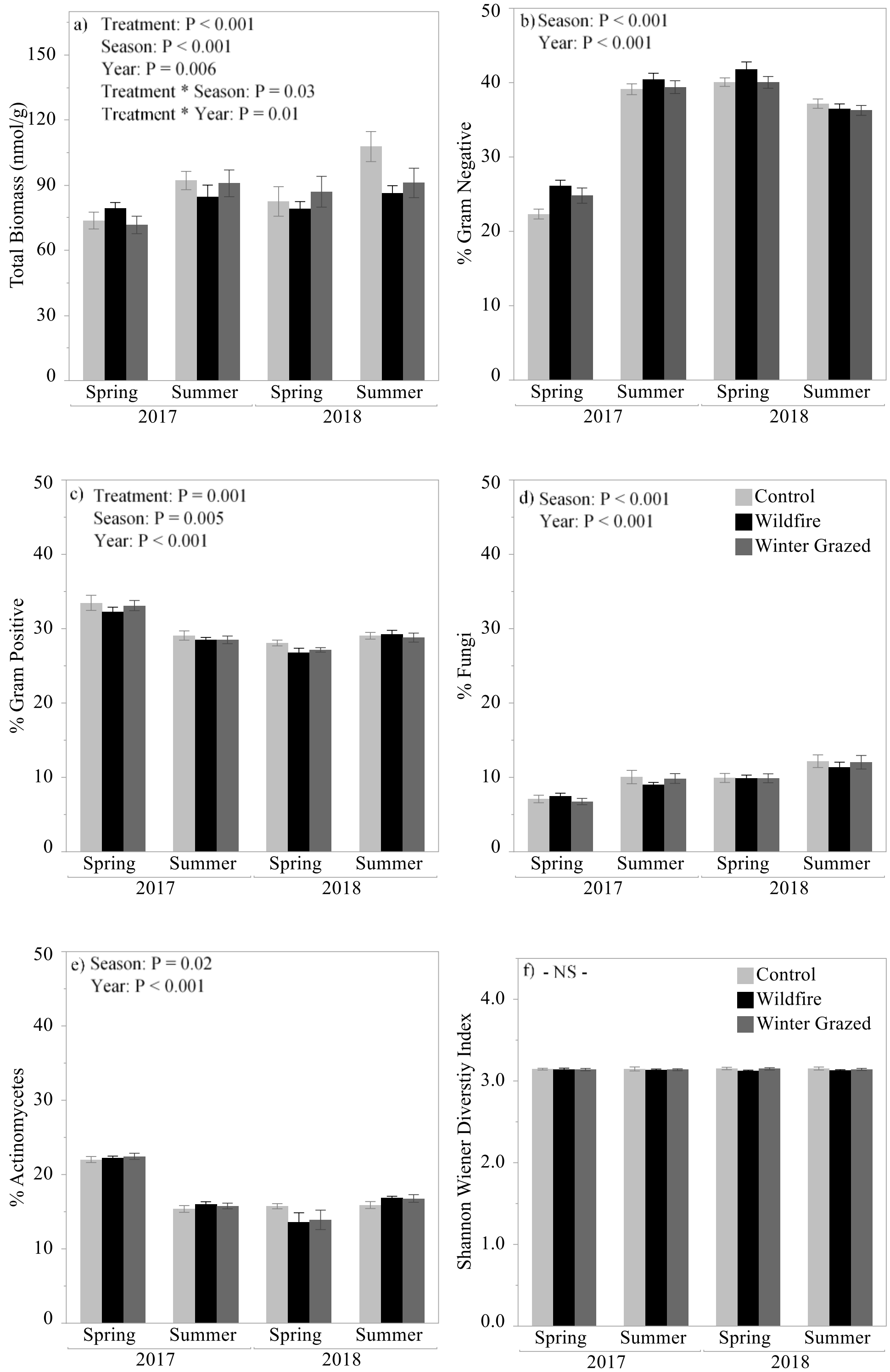


Figure 2:

Mean total biomass, percent gram-negative bacteria, percent gram-positive bacteria, percent fungi, percent actinomycetes, and the Shannon Wiener diversity index of each season within each year, in the mixed-grass block. Significant model effects are indicated by their p-value. -NS- indicates no model effects were significant.

ShortGrass Block

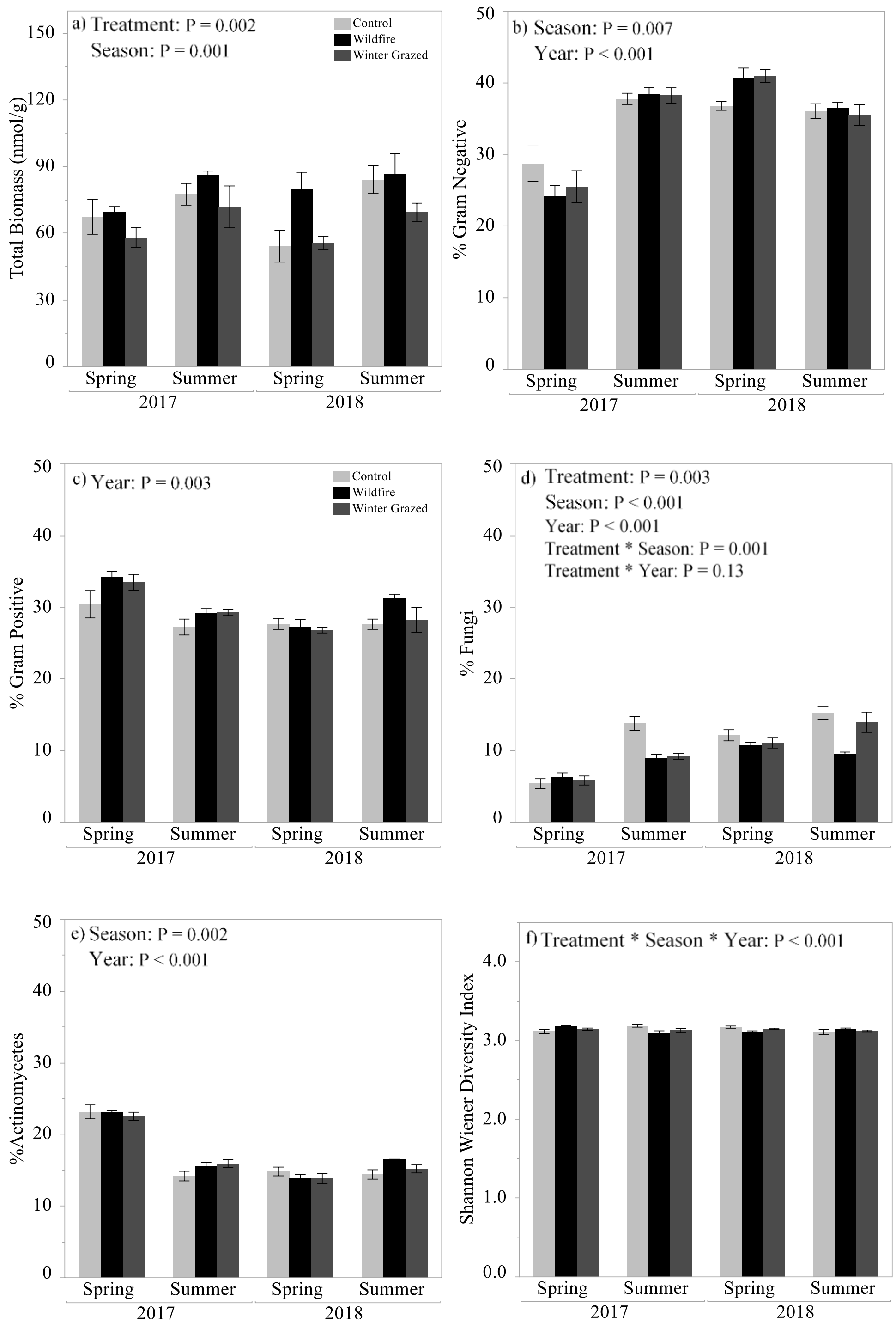


Figure 3:

Mean total biomass, percent gram-negative bacteria, percent gram-positive bacteria, percent fungi, percent actinomycetes, and the Shannon Wiener diversity index of each season within each year, in the shortgrass block. Significant model effects are indicated by their p-value. -NS- indicates no model effects were significant.

CHAPTER 3: EFFECTS OF GRAZING AND FIRE ON HYDROLOGICAL PROCESSES IN THE NORTHERN GREAT PLAINS GRASSLAND

Abstract

Grazing and fire, in the Northern Great Plains grasslands, can influence soil hydrologic processes including soil moisture and temperature, infiltration rates, surface runoff, soil loss, and sediment yield. This study assessed the impact of summer-long grazing (CG), high-intensity winter-grazing (WG) and wildfire (WF), with three different hillslopes (2%, 3%, and 5%), on surface runoff, soil loss, and sediment yield in western South Dakota grasslands, using measured field data and the watershed erosion prediction project (WEPP) model. Soil moisture and temperature sensors were installed at three depths (6-, 12, and 24-inch), across replicates of three land surface disturbance areas (3 CG, 3 WG, and 3WF pastures), and monitored for approximately two years (April 13th, 2017 to December 31st, 2018). Infiltration tests were also conducted within each land surface disturbance area every two months, weather permitting, from June 2017 to August 2018. For the WEPP model, a total of nine scenarios were constructed for a period of January 1st, 2017 to December 31st, 2018 and compared with a baseline scenario consisting of summer-long grazing that occurred on a 2% hillslope. The scenarios were designed to evaluate land surface disturbance individually, hillslope individually, and the combined effect of land surface disturbance and hillslope. Field assessment showed that soil moisture and temperature were affected by land surface disturbances. Control grazing had the highest soil moisture content. The effect of winter-grazing and wildfire on soil moisture varied by depth, with winter-grazing having higher soil moisture than wildfire at

the 6-inch depth and lower soil moisture than wildfire at the 12-inch depth. Wildfire had the highest soil temperature followed by winter-grazed disturbance. However, these observed differences in soil moisture and temperature were not likely to cause any biologically significant effects. Although winter-grazed show the longest infiltration times followed by that of the summer-long grazing, infiltration rates between land surface disturbance were not significantly different. During the modeling exercise, when only land surface disturbance is altered from the baseline, winter-grazing does not lead to increased surface runoff, soil loss, or sediment yield when compared to the baseline scenario. The wildfire scenario, however, increases runoff by 25%, soil loss by 444%, and sediment yield by 421%. As hillslope increased, surface runoff increased slightly from 1 to 60% in contrast to soil loss and sediment yield. Soil loss and sediment yield increased by as much as 340 to 1,100% and 200 - 700% respectively, from summer-long grazing (baseline). The combined impacts of land surface disturbance and hillslope have a minimal effect on winter-grazed scenarios but amplify the effect of wildfire scenarios.

Introduction

Grazing and fire are two prominent land surface disturbance strategies in the Northern Great Plains (NGP) that influence soil moisture and temperature, infiltration rates, surface runoff, soil loss, and sediment yield (Hubbard et al. 2004, Shaw 2005, Teague et al. 2008, Teague et al. 2010). Land surface disturbance strategies such as grazing and wildfire and their location on a hillslope should be better understood in regard to their effect on erosion processes; which impacts ecosystem services provided by rangelands (Assessment 2005).

Since European settlement, wildfire frequency has been reduced. Plant litter, that serves as the fine fuel responsible for wildfires, was reduced as livestock grazing increased (Bock et al. 1993). Increases in grazing also reduced the frequency of fire by embedding plant litter into the soil, making it unavailable to burn (Nader et al. 2007). During the last 150 years, fire, however, has also been reduced due to liability issues, forage concerns, and ranchers not having the knowledge and equipment to perform prescribed burns on their lands (Toledo et al. 2014). Along with fire, grazing regimes have shifted from large herds of bison that would graze on recently burned rangelands, to prescribed grazing that promotes vegetation most beneficial for cattle production (Fuhlendorf and Engle 2001). Due to these changes to fire and grazing regimes that decrease rangeland benefits, alternative land surface disturbance strategies, that include grazing and fire, are needed to improve rangeland functionality. As alternative land surface disturbance strategies are developed to combat the loss of diversity, their effect on surface runoff, soil loss, and sediment yield, in relation to their location on a hillslope, should be evaluated for their potential to serve as a sustainable land surface disturbance strategy.

Fire and livestock grazing have the potential to increase rangeland diversity but their impact on erosion processes, which affect ecosystem services, should be first understood. Fire in the NGP increases the initial risk of water erosion as a result of reduced plant cover (Teague et al. 2008, Teague et al. 2010). However, fire can serve as a surface disturbance tool to increase rangeland plant richness, which can improve hydrologic processes over time (Briske 2017). For example, grassland species richness was found to increase two years post-fire in California (Harrison et al. 2003). Along with increased richness, the removal of litter enhances the availability of nutrients and increases grassland plant productivity (Parsons and Stohlgren 1989). Increases in plant species richness and productivity led to a reduction in the amount of erosion that occurred during rainfall simulation experiments on square meter plots in New Mexico (Wilcox and Wood 1989).

With 70% of lands available in the western United States being used for livestock grazing, livestock grazing is the most prominent land management activity (Bock et al. 1993, Fleischner 1994). Since livestock grazing became a dominant land management activity across NGP rangelands, water quality in these areas has decreased due to increases in erosion and sedimentation (McGinty et al. 2009). Research showed that as grazing intensity increases, the amount of bare ground increases resulting in greater amounts of erosion and sedimentation (Gill et al. 1998).

Livestock can affect water quality directly and indirectly by increasing sedimentation and nutrient loading, promoting the spread of enteric pathogens, and raising water temperature (Doran and Linn 1979, George and Clawson 1993, Campbell and Allen-Diaz 1997, Hubbard et al. 2004). However, if correctly implemented, livestock grazing can maintain or improve erosion processes such as surface runoff and soil loss, by increasing rangeland plant diversity and productivity (Blackburn 1983). Different degrees of stocking rates (light to intense) and timing of implementation are factors that affect the amount of erosion that occurs on a field. Intense grazing can decrease soil loss if implemented at low frequencies (Briske 2017). Fuhlendorf and Engle (2001) showed that intense grazing can improve rangeland biodiversity and plant productivity when implemented periodically. Increases in plant productivity decrease surface runoff and reduce erosion rates (Wilcox and Wood 1989).

When evaluating the effectiveness of land surface disturbances, that involve fire and livestock grazing, topographic characteristics, should be considered, as they influence the amount of erosion that occurs. In general, as slope increases, erosion also increases (Fox and Bryan 2000). However, the amount of erosion that increases, as a result of increased slopes, can be amplified if an intense land disturbance is implemented. Thus, alternative land surface disturbance strategies may not be suitable across all topographic conditions (Evans 1998). The limiting slope for a sustainable grazing practice has been demonstrated across various ecosystems (Evans 1996, 1997, Evans 1998). On rangelands in upland England, on slopes greater than 26%, hillslopes have a greater risk of becoming overgrazed and exposing bare ground (Evans 1997). In Norway, Reindeer have increased risk of exposing bare ground when slopes exceed 7% (Evans 1996). Once bare ground is exposed due to overgrazing, which can be amplified by hillslope gradient, vegetation will have difficulty stabilizing and recolonizing (Evans 1998). Although these examples are from areas that differ from the NGP, they illustrate the importance to evaluate the impact of a land disturbance strategy on hillslopes of specific areas of interest.

In this study, soil moisture, temperature, and infiltration rates were monitored to evaluate the impacts of grazing and fire. Surface runoff, soil loss, and sediment yield were also modeled using the Watershed Erosion Prediction Project model (WEPP) which has been used to assess the impacts of grazing and fire on runoff and erosion processes. The WEPP model is capable of quantifying the impacts of grazing and fire on surface runoff, soil loss, and sediment yield from hillslope scales (Larsen and MacDonald 2007, Pieri et al. 2007, Quinn 2018). For example, Quinn (2018) used climate and land use data, along with field data to evaluate the impact of fire on erosion processes such as surface runoff and sediment yield with the WEPP model. The present study followed similar approaches and used climate and land use data, along with field data at the Cottonwood Field Station, in Cottonwood South Dakota. The specific objective of this study was to quantify the individual and combined effects of land surface disturbances and hillslopes. Specifically, grazing during the winter months at an extremely high intensity and wildfire were evaluated on three different hillslopes for surface runoff, soil loss and sediment yield at the Cottonwood Field Station.

Methods

Study Area

This study was conducted on three hillslopes at the Cottonwood Field Station (43°96'07" N, 101°85'81" W; Figure 1). The Cottonwood Field Station is approximately 120 km east of Rapid City, South Dakota. The landscape at Cottonwood is dominated by the Northern Great Plains mixed-grass prairie on gently sloping long rolling hills with relatively flat-topped ridges. According to the National Land Cover Database (NLCD 2016; Yang et al., 2018), all hillslopes were completely dominated by grassland/herbaceous cover. Major vegetation includes western wheatgrass (*Pascopyrum smithii*), green needlegrass (N*assella viridula*), buffalograss (*Bouteloua dactyloides*), and blue grama (*Bouteloua gracilis*). The average annual precipitation recorded between 2017 and 2018 at the Cottonwood Field Station was 31.1 cm (12.2 inches). The climate is characterized as semi-arid and continental. Based on information extracted from the Web Soil Survey (WSS, Soil Survey 2018) the dominant hydrologic soil group at the Cottonwood Field Station is "D", with 26% silt in the top 100 cm (39.4 inches) of the soil profile (Mesonet 2019). Historically, cattle grazing with low/moderate stocking rates has been the major practice at the Cottonwood Field Station for more than 70 years.

Description of Land Surface Disturbance

Three soil surface disturbance treatments including summer-long continuous grazing (CG-control), a single high-intensity winter-grazing event (WG), and wildfire (WF) were evaluated for their effects on soil temperature, soil moisture, and infiltration rates. The treatments were implemented with a randomized complete block design in three separate pastures (Fig. 2). The three pastures ranged in size from 60 ha (130 ac.) to 73 ha (180 ac.). Prior to winter-grazing or wildfire treatments, all three pastures were summer-long grazed in 2016 with approximately 1.04 animal unit months/ha (AUM/ha). Following the summer-long grazing, in 2016, a high-intensity wildfire burned the southern portions of all 3 pastures in October of 2016, prior to the implementation of winter-grazed surface disturbance treatments. Each pasture had approximately a third of their area burned by the wildfire. The area burned by the wildfire was then fenced and not grazed further. Another third of each pasture was winter grazed once at a high intensity. To implement winter-grazed treatments a single herd of 120 steers was used to graze the area for approximately 30 days until vegetation was reduced to an approximate height of 8-10 cm. Once the vegetation was reduced to approximately 8-10 cm, the herd was removed and immediately placed on another winter-grazed area. All grazing times occurred during the winter months (December – February) and had an approximate stocking rate of 1.62 AUM/ha. The third of each pasture that was only grazed with summer-long grazing at 1.04 AUM/ha in 2016 and did not experience either the wildfire or winter grazing and was used for the control. All three treatment areas had a portion of their perimeter adjacent to the other two treatment areas. Sampling exclosures (total 9 exclosures) made of 4-gauge cattle panels that were approximately 24 square meters were placed in the corner of each treatment area in close proximity to the other two treatment areas (Fig. 2.). Exclosures placement reduced environmental variability and prevented further grazing from livestock and damage to sampling equipment.

Field Data Collection

Soil moisture and temperature were recorded every 15 minutes using ECH$_2$O 5TM soil probes (METOS, Werksweg, Austria). ECH$_2$O 5TM soil probes were placed in each exclosure within each treatment area at 6-, 12-, and 24- inch depths. A total of 27 soil moisture probes were used altogether in the study across the three treatments and three depths. Fifteen-minute temperature and volumetric water content (VWC) data were aggregated into daily data from April 13th, 2017 to December 31st, 2018. Soil moisture and temperature data were used at the daily timescale in all analyses.

Infiltration rates were measured using a single ring infiltrometer following the procedures discussed by Herrick et al. (2005). Three randomized sampling sites were selected within each exclosure (Fig. 2). The above-ground litter was removed, without disturbing the soil crust, and a single ring infiltrometer was driven into the ground until approximately 2 inches of the ring remained above the surface. The ground surface area inside the infiltration ring was covered with a plastic barrier (saran wrap). 500 mL of water was slowly poured onto the plastic barrier (approximately 1 inch of water). The barrier was then gently removed at once and the amount of time it took for the water to infiltrate the soil was recorded. Infiltration was considered complete when no standing water remained on the surface of the ground inside the ring. The samples were collected approximately every two months from June 2017 through August 2018, weather permitting. A total of 189 infiltration measurements were taken, 63 per soil surface disturbance treatment.

Statistical Analysis of Field Data

All data were analyzed with SAS (JMP Pro, Version 11. SAS Institute Inc., Cary, NC, 1989-2007) and R (R Core Team (2013)). Assumptions of normality, for soil temperature, moisture, and infiltration rates were tested using the Shapiro-Wilk test (Shapiro/Wilk). Soil moisture, temperature, and infiltration rates residuals failed to meet the assumption for homogeneity of variance and an analysis to determine if soil moisture, temperature, and infiltration rates medians, between treatments, differed from one another was conducted using the non-parametric Kruskal-Wallis test. Where significant differences in soil moisture, temperature, and infiltration rates existed, between treatment medians, a post-hoc non-parametric comparison for each pair was performed using Wilcoxon Method test. This allowed comparisons of median soil moisture, temperature, and infiltration rates between treatments.

WEPP Model

The Water Erosion Prediction Project (WEPP) Windows-based interface (Version 2012.8) was used to simulate the impacts of two disturbances (grazing and wildfire) on runoff and soil loss on various hillslopes. WEPP is a process-oriented, continuous simulation, erosion prediction model capable of assessing the impact of land surface disturbance practices on runoff, soil loss, and sediment yield at hillslope and sub-watershed scales (Flanagan and Nearing 1995). The WEPP model includes several conceptual components which are used to estimate soil erosion processes. These components include: climate (precipitation, wind speed/direction, solar radiation, evaporation) including winter (freeze-thaw, snow accumulation, snow melting), irrigation scheduling, hydrology (infiltration, depressional storage, runoff), water balance (evapotranspiration, percolation, drainage), soils (types and properties), crop growth (cropland, rangeland, and forestland), residue management and decomposition, tillage impacts on infiltration and erodibility, erosion (interrill, rill, channel), deposition (rills, channels, and impoundments), and sediment delivery (Flanagan et al. 1995). WEPP has the capability to delineate hillslopes within sub-watersheds, which can then be isolated and evaluated for runoff, soil loss, and sediment yield at the hillslope scale. It has been widely applied on many watersheds, hillslopes, and geographic locations, along with varying land disturbance activities such as different intensities of grazing and fire (Duiker et al. 2001, Larsen and MacDonald 2007, Pieri et al. 2007).

WEPP Input Data

To analyze the impacts of grazing and wildfire on runoff, soil loss, and sediment yield, on three hillslopes at Cottonwood Field Station, a 'baseline" model was created for a 2-year study period, from January 1st, 2017 to December 31st, 2018. The input data for the WEPP model were first created in the Geospatial interface for the WEPP model (GeoWEPP, version ArcGIS 10.4.0). GeoWEPP utilizes digital geo-referenced information, such as digital elevation models (DEM) and topographical maps, to derive and prepare model input parameters for assessing erosion processes for a small watershed with a single soil and land use (Flanagan and Nearing 1995). Input parameters created within GeoWEPP were exported to WEPP Windows-based interface to be evaluated at the hillslope scale. Creation of the WEPP model in the GIS interface (GeoWEPP) requires topography, soil texture, land cover, initial conditions, and climate data. The input data were extracted from a variety of sources. To extract topographic data, a 30 m DEM was obtained from the South Dakota Department of Environmental and Natural Resources database (DENR 2015). The DEM was used to determine the three hillslope profiles in the study area (Figures 3-5). The three hillslope profiles delineated resulted in 2%, 3%, and 5% slopes. As discussed by Flanagan and Nearing (1995), topographic data imported into GeoWEPP undergo rectangular hillslope abstraction. This prevents the model from incorporating lateral curvature across a hillslope, giving all estimates per unit width at distances down the hillslope.

30 m resolution, land use data were obtained from the National Land Cover Database (NLCD) for the year 2016 (DENR 2015). From this land-use dataset, rangeland plant parameters were altered based on long-term historical data (Table 1). WEPP rangeland plant parameters from the model's rangeland growth component is a modification of the EPIC model's crop growth model and accounts for water and temperature stresses on biomass production (Arnold et al. 1995). As suggested in the

WEPP user manual, caution was taken when adjusting the rangeland plant parameters by changing parameters to reflect the field conditions based on historical data collected at the Cottonwood Field Station and expert knowledge of rangeland plant communities at the site. Adjustments to rangeland plant parameters are normally used for sensitivity analysis purposes when rangeland plant growth and condition is of interest after disturbance (Arnold et al. 1995). Because simulations were not focused on the effects of different plant communities on runoff and soil loss, rangeland plant conditions were left constant across all simulation scenarios as discussed in a section below (Table 1).

Soil data were obtained from WSS at a scale of 1:24,000. Soil properties obtained from WSS were used as initial soil parameters but changed later during calibration of the model.

Initial conditions (see Table 2) were also required for WEPP model simulations (Arnold et al. 1995). Initial conditions are the field conditions that exist on January 1st of the first simulation year (2017). During continuous model simulations, the effect of initial conditions on model output is minimal due to the simulated plant growth and decay as well as climate inputs that greatly affect the initial conditions throughout the simulation period (Flanagan et al. 1995).

A single weather file was created using observed total daily precipitation, minimum and maximum daily temperature, and daily solar radiation obtained from the South Dakota Mesonet site (Mesonet 2019), located at the Cottonwood Field Station. All other related climate components (e.g. solar radiation, wind direction, wind speed, dew point, rainfall duration,) were developed through a stochastic weather generator (Nicks et al. 1995) within WEPP. The CLIGEN climate generator produces individual storm parameter estimates for each rainfall event based on long-term historical data (USDA 2016).

Model Calibration and Validation

A baseline WEPP model was constructed using the digital elevation model, NLCD 2016, climate data, and initial conditions (i.e. conditions that existed on January 1st, 2017 at the beginning of the study; see Table 2 and Input Data section). The baseline model represents control grazing on 2% hillslope. The model was set up for this project with rangeland option (Flanagan and Nearing 1995) and calibrated and validated for daily soil moisture from January 1st, 2018 to December 31st, 2018 and from April 13th, 2017 to December 31st, 2017, respectively. The coefficient of determination (R^2) and the Nash-Sutcliffe Efficiency (NSE) statistics were used to evaluate the performance of the model. R^2 describes the proportion of the variation explained by the model and range in value from 0 to 1 (Nagelkerke 1991). Values close to 1 indicate that the model explains a higher proportion of the data variability. NSE statistic is also frequently used to assess the performance of hydrological models (Nash and Sutcliffe 1970). NSE has an upper limit of 1 which demonstrates perfect alignment with observed data. The WEPP model is deemed to have satisfactory simulations if the R^2 and NSE values are ≥ 0.5, whereas values below 0.2 are likely to be considered insufficient (Moriasi et al. 2007).

In this study, we followed sequential parameterization and manual calibration for soil parameters of the control grazing pasture hillslope defined in this study. WEPP parameters and their initial value ranges were selected based on soil data imported from WSS. The baseline scenario grazing intensity and hillslope are described in the section below. As mentioned earlier, observed soil moisture data were used for model calibration and validation. The soil moisture data were collected in the field with ECH_2O 5TM soil probes (METOS, Werksweg, Austria). Measurement uncertainty for soil moisture was assumed negligible since the soil moisture data were obtained from calibrated soil moisture probe readings.

For calibration of the WEPP model, a sensitivity of analysis was performed on parameters representing soil properties of the control pasture (see Table 3) by manually modifying individual parameters to determine parameters that were most sensitive to changes in soil moisture. The most sensitive parameters were detected by individually manually increasing the default parameter values in the model by 100% to determine changes in soil moisture. Soil depth, percent sand, and percent clay in the soil were most sensitive to cause substantial changes in soil moisture. Once the most sensitive parameters were determined, modifications of the more sensitive parameters were further adjusted manually and individually, at 1% intervals from the initial values, in attempts to obtain satisfactory R^2 and NSE values. Parameters used for calibration of the model were not modified more than 20 percent and always increased from the default values (Table 3). After obtaining the best estimates of the soil parameters for calibration, the baseline model was validated for the same location but for a non-overlapping period from April 13th, 2017 to December 31st, 2017.

Implementation of Land Surface Disturbance in WEPP

Three different land surface disturbances were simulated in this study. They consist of conventional summer-long grazing (baseline), a single winter grazing event that occurred for one continuous month (January 1^{st} through February 1^{st}) at an extremally high intensity (WG), and a wildfire with an extremally high intensity (WF) that occurred on October 15^{th} of the first simulation year (2017)

The baseline scenario or the summer-long grazing was simulated by applying a 'grazing schedule' to the WEPP model during the first and second simulation year (Table 4). For the first simulation year (2017) 22 cattle, with an average weight of 700 lbs. were grazed on 162 acres(65 ha) from the 15^{th} of May to the 13^{th} of July. In the second simulation year (2018) 19 cattle, with a weight of 700 lbs., were grazed from the 15^{th} of May to the 15^{th} of August. A drought that occurred in the first simulation year (2017) resulted in a shorter grazing period than the second simulation year (2018). Grazing characteristics that are required to implement grazing with the model include (a) maximum digest of forage = 0.5 (b) minimum digest of forage = 0.2 and (c) fraction of forage available = 0.8. The baseline model represents the control land surface disturbance and hillslope. The baseline model in this study was defined as conventional grazing (i.e. summer-long grazing without rotation) on a 2% slope.

Winter-grazing (WG) was simulated by adding a grazing schedule to the baseline simulation from the 1^{st} of January to the 1^{st} of February with 120 cattle that averaged 750 pounds (Table 5). All other parameters from the baseline simulation were left constant. Changes to maximum or minimum digest of forage were not altered, as no attempt was made to predict specific vegetation characteristics for scenario simulation. It should be noted that the grazing schedule and cattle number on the pasture for the baseline and winter grazing scenarios were constructed to replicate the surface disturbance practices that occurred in the field at the Cottonwood Field Station.

The version of the WEPP model (version 2012.8) used in this study accepts 'burning' as a land management option for rangeland, and thus this option was selected for the wildfire disturbance simulations (WF) (Table 6). To simulate wildfire, five field input data consisting of (a) live biomass fraction accessible for consumption following burning (value entered in the model- 0), (b) fraction reduction in standing wood mass due to burning (value entered in the model-1), (c) fraction change in potential above-ground biomass (value entered in the model- 0), (d) fraction evergreen biomass remaining after burning (value entered in the model- 0), and (e) fraction non-evergreen biomass remaining after burning (value entered in the model- 0). The values used for the simulations represent a wildfire of the highest severity.

Simulation of Land Surface Disturbance with WEPP

Nine total simulations were performed in this research, including the baseline scenario. As stated above, the baseline scenario consists of conventional summer-long grazing with no rotation on a 2% slope. All input parameters (initial conditions, plant characteristics, and soil properties) were left consistent with the baseline scenario when other land surface disturbance scenarios were implemented, or hillslope changed. To develop alternative scenarios, we first modified the land surface disturbance from the baseline scenario. These scenarios consisted of winter grazing and wildfire land surface disturbance scenarios occurring on the 2% slope. Next, the hillslope of the baseline was altered to 3% and 5% slopes. These slopes were true hillslopes of the Cottonwood Field Station. Finally, we modified land surface disturbance and slope of the baseline to obtain the winter grazing on hillslope 3% and hillslope 5%, and wildfire on hillslope 3% and hillslope 5%. These changes from the baseline result in 8 different scenarios in addition to the baseline scenario (Table 7).

Results

Field Assessment

Median soil moisture content was statistically significant and affected by surface disturbance treatments at the 6-inch ($p < 0.001$), 12-inch ($p < 0.001$), and 24-inch depths ($p < 0.001$) (Fig. 6). Control treatments had the highest soil moisture across all depths, followed by winter grazed. Wildfire at the 6-inch depth had the lowest soil moisture. However, the differences in soil moisture, between treatments, across all three depths, are not likely to have any biological significance. At the 6-inch depth the difference between the land surface disturbance treatment with the highest soil moisture content, CG, and the land surface disturbance treatment with the lowest soil moisture content, WF, is only 0.04 m^3/m^3. At the 12-inch depth, the greatest difference between land surface disturbance treatments is 0.05 m^3/m^3. The greatest difference in soil moisture, between surface disturbance treatments, is at the 24-inch depth, with a 0.06 m^3/m^3 difference, which is not likely to be large enough to cause any biological effect. In general, water stress begins to occur when VWC has depleted by 20% for clay soils and 8% for extremely sandy soils, from field capacity (Pitts 2016). In this study, the decrease between 4% to 6% is not likely to cause any biological effect on plant productivity.

Median soil temperature was also statistically significant and affected by land surface disturbance treatments at the 6-inch ($p = 0.02$), 12-inch ($p = 0.02$), and 24-inch depth ($p < 0.001$) (Fig.7). Across all depths, wildfire had the highest soil temperature, followed by winter-grazed and summer-long grazing. Similar to soil moisture, the greatest difference in soil temperature, between surface disturbance treatments, is not likely to be considered biologically significant. The greatest differences between the surface disturbance treatments range from 0.6 °C to 0.9 °C, depending on depth. These differences in surface disturbance treatments, although statistically significant, are not biologically significant.

Infiltration rates were not significantly affected by land surface disturbance treatments ($p = 0.33$) (Fig. 8). Mean infiltration rates varied from 5 to 8 minutes between treatments. Individual infiltration rates collected in the field varied greatly with respect to the time it took to absorb the water. This high variability in the infiltration rates results in high uncertainty in the effects of the surface disturbance treatments.

WEPP Model Performance

The baseline model which is the summer-long continuous grazing on 2% hillslope was manually calibrated and validated with 11 parameters for soil moisture. The model performance statistics show R^2 and NSE values greater than 0.5 for daily and monthly soil moisture (Table 8; Fig. 9). The performance of the model for the entire simulation period (January 1st, 2017 – December 31st, 2018) also shows satisfactory statistics as presented in Table 8. Based on suggested model calibration guidelines (Moriasi et al. 2007), these performance statistics were deemed satisfactory for the proposed modeling exercise in this study. There was no distinct pattern between observed and simulated soil moisture over the calibration and validation periods; however, the model tended to overestimate soil moisture toward the end of the dryer year of 2017 and slightly underestimated soil moisture in the growing season of 2018 (see Fig. 9). The overestimate of soil moisture during the dry year of 2017 could be attributed to the drought that began toward the end of June 2017 and lasted through the latter parts of October, and into November 2017. Since the model was calibrated with a wetter year (January 1^{st,} 2018-December 31^{st,} 2018), the adjusted parameters were likely not able to capture drought processes during the year 2017, leading to the slight overestimation of soil moisture in 2017. After the drought in 2017, at the beginning of 2018, the model underestimated soil moisture slightly.

Baseline Scenario

For the baseline scenario, annual surface runoff (Fig. 10) across both project years was 2.5 cm (1.0 in). Soil loss for the baseline scenario (Fig 11), had an average of 40.4 kg/ha/yr (0.018 ton/A/yr). Sediment yield (Fig. 12), in the baseline scenario, was similar to soil loss over the entire study period with an average of 42.6 kg/ha/yr (0.019 ton/A/yr).

Land Surface Disturbance Effect

Land disturbance was simulated by applying two different land surface disturbance scenarios to the baseline scenario. These are winter grazing at an extremely high intensity (WG) and a wildfire (WF). Results reveal that the WG land surface disturbance scenarios resulted in similar amounts of surface runoff, soil loss, and sediment yield as the baseline, while wildfire resulted in greater amounts of surface runoff, soil loss, and sediment yield than the baseline, across both project years (i.e. 2017 and 2018). WG land surface disturbance scenarios had an average of 2.5 cm/yr (1 in/yr) of surface runoff (Fig. 10), a 1% increase from the baseline scenario, while WF had 3.2 cm/yr (1.2 in/yr) of runoff, a 25% increase from the baseline scenario. WG land surface disturbance scenarios did not increase the amount of soil loss (Fig. 11) compared to the baseline scenario with 40.4 kg/ha/yr (0.018 ton/A/yr). WF increased the amount of soil loss by 444% compared to the baseline scenario, with 219.7 kg/ha/yr (0.098 ton/A/yr). Sediment yield, for WG land surface disturbance scenarios, did not increase from the baseline and had 42.6 kg/ha/yr (0.019 ton/A/yr) of sediment yield (Fig. 12). WF increased the amount of sediment yield from the baseline by 421% with 221.9 kg/ha/yr (0.099 ton/A/yr).

Slope Effect

Topography was altered by selecting three hillslopes (2%, 3%, and 5%, Fig. 3 - 5) at the Cottonwood Field Station. Results reveal that surface runoff does not increase as sediment yield and soil loss do with increasing slope. As hillslope is increased to 3% the amount of runoff (Fig. 4) increases by 1% or 2.5 cm/yr (1.0 in/yr) compared to the baseline scenario. The increase in runoff is greater as the hillslope increases to 5%, with 61% increase or 4.0 cm/yr (1.6 in/yr) of runoff occurring. Unlike surface runoff, soil loss increases as the hillslope increases from the baseline (2%) to 3% (Fig. 11). At the 3% hillslope soil loss increases 344% from the baseline with 179.3 kg/ha/yr (0.08 ton/A/yr) of soil loss occurring. As the hillslope increases further to 5%, 479.7 kg/ha/yr (0.21 ton/A/yr) of soil loss occurs, representing 1,089% increase from the baseline. The amount of sediment yield that occurs in the simulations also increases significantly as hillslope increases from the baseline (2%) (Fig. 12). On 3% hillslope, 125.5 kg/ha/yr (0.056

ton/A/yr) of soil loss occurs, which is a 195% increase from the baseline scenario. As hillslope increases to 5%, 354.2 kg/ha/yr (0.16 ton/A/yr) of sediment yield occurs, a 731% increase from the baseline.

Land Surface Disturbance *and Slope Effect*

As changes occur in slope and land surface disturbance their combined effects, across both project years (2017 and 2018), increase the amount of surface runoff, soil loss, and sediment yield. In other words, the amount of surface runoff, soil loss and sediment yield increases caused by land-surface disturbance scenarios would be enlarged with increases to hillslope gradients. Amplified changes occur most drastically in soil loss and sediment yield with less dramatic changes in surface runoff, compared to the baseline. Surface runoff for winter grazed land surface disturbance scenarios are predicted to increase by 1% to 2.5 cm/yr (1in/yr) and 67% to 4.2 cm/yr (1.7 in/yr) on 3% and 5% hillslopes, respectively (Fig. 10). WF surface runoff is expected to increase 28% to 2.5 cm/yr (1.3 in/yr) and 88% to 4.7 cm/yr (1.9 in/yr) on hillslope 3% and 5% respectively. Soil loss (Fig. 11) for WG is expected to increase 344% to 179.3 kg/ha/yr (0.08 ton/A/yr) and 1,117% to 490.9 kg/ha/yr (0.22 ton/A/yr) on 3% and 5% hillslopes, respectively. Soil loss for WF increases dramatically more on hillslope 3% and 5% with a 1,438% increase to 621.0 kg/ha/yr (0.28 ton/A/yr) and a 3,094% increase to 1,289.0 kg/ha/yr (0.58 ton/A/yr). WG sediment yield also had dramatic increases from the baseline on 3% and 5% hillslopes with a 195% increase or 125.5 kg/ha/yr (0.056 ton/A/yr) and a 747% increase or 360.9 kg/ha/yr (0.16 ton/A/yr) increase from the baseline scenario (Fig. 12). WF increases sediment yield more that WG on hillslope 3%

and 5% with a 926% increase or 437.1 kg/ha/yr (0.20 ton/A/yr) and a 2,210% increase or 984.1 kg/ha/yr (0.44 ton/A/yr).

Discussion

The results of the field data analysis show that soil moisture and temperature were affected and infiltration rates were not affected by land surface disturbance treatments. Although there were differences between treatments for soil moisture and temperature, these differences were very small and not likely to have any biological effect. Control treatments had the highest soil moisture, possibly as a result of more litter cover. Unlike the wildfire or winter grazed treatments, which removed litter cover through burning or grazing, summer-long grazing treatments maintained litter cover. By maintaining litter cover, soil moisture does not decrease as rapidly when compared to surface disturbance treatments that decrease litter cover, such as fire (Hulbert 1969). However, the regeneration of plant biomass can restore litter and result in no differences after the first growing season following disturbance (Hulbert 1969). Soil temperature varied from 0.6 °C to 0.9 °C, depending on depth, a difference that is not considered to be biologically significant in relation to plant production. Vermeire et al. (2005) also studied the response of soil temperature following fire and found 1 - 3 °C increases in soil temperature following burning. The difference in soil temperature found by Vermeire et al. (2005), after burning, was also not considered to have any biological effect. surface disturbance treatments had no effect on infiltration rates in this study. Infiltration measurements taken during this study were highly variable and may limit the interpretation of the surface disturbance treatments effects.

Results from our modeling showed that winter-grazed land surface disturbance scenarios do not drastically increase the amount of surface runoff, soil loss, or sediment yield when compared to summer-long grazing. In other studies, surface runoff amounts have been found to increase with increases in grazing intensity (Rauzi and Hanson 1966). However, these increases in surface runoff are expected after long-duration or continuous intense grazing over several years. In the present study, the intense stocking rate, from the winter grazed scenarios, only occurred once and only for a short duration, which resulted in less surface runoff (Figure 6). This is similar to results found by Sanjari et al. (2010) when pastures are grazed under short durations with sufficient rest periods, ground cover improved and surface runoff did not increase. Similar to runoff, soil loss for the winter grazed scenario did not increase when compared to the baseline. The limited soil loss for winter grazed land surface disturbance scenarios is directly linked to the amount of runoff that occurred for the scenarios simulated. As surface runoff increases the amount of soil loss increases as well (DiBiase and Whipple 2011). Based on this relationship between runoff and soil loss, the amount of soil loss in winter grazing scenarios could not be expected to increase (DiBiase and Whipple, 2011), since no increase in surface runoff was observed. The strong correlation between sediment yield and soil loss in our simulation is expected as sediment yield is directly related to soil loss in the WEPP model when an impoundment is not located along the hillslope (Flanagan and Nearing 1995). Sediment yield after an intense grazing event has been demonstrated to increase on a hillslope, but only when implemented on a rotational basis (Warren et al. 1986). The winter-grazed land surface disturbance scenario in this study, however, was not implemented on a rotational basis but rather over a short duration (January 1st, 2017 to February 1st, 2018). This short duration, intense grazing, could not increase sediment yield, similarly the findings in the study by Sanjari et al. (2010) which, showed that intense grazing events that occur during a short time period did not increase sediment yield.

In contrast to our winter grazed land surface disturbance scenario, our wildfire land surface disturbance scenarios did create a large increase in surface runoff, soil loss, and sediment yield, from summer-long grazing (Fig. 6-8). This is expected as the amount of runoff that occurs in the immediate years following a wildfire is significantly greater when compared to areas that have not experienced a burn, due to the reduction of standing vegetation (Larsen et al. 2009). As surface runoff increases, the amount of soil loss also increases (DiBiase and Whipple 2011). The wildfire scenarios had substantially more surface runoff than summer-long grazing scenarios and are therefore expected to increase the amount of soil loss that occurs on a hillslope. The dramatic reduction of standing cover also increases the amount of soil loss and sediment yield that occurs after a wildfire. When standing cover is reduced, and 60-70% of bare ground is exposed (Johansen et al. 2001), the amount of soil loss resulting from a fire increases.

In addition to the impacts of land surface disturbance scenarios, increases to hillslope gradients would increase the amount of runoff, soil loss, and sediment yield (Fig 6-8). Runoff in the simulated scenarios increased noticeably when the hillslope increased to 5%. There was a very minimal increase when hillslope only increased by 1% from the baseline to the 3% hillslope. As hillslope increases, the amount of runoff that occurs is expected to increase (El Kateb et al. 2013). The lack of a noticeable increase for the baseline between 2% slope and the 3% slope could result from only the slight increase in hillslope gradient, which was not large enough to cause notable increases in runoff. The increases in the amount of soil loss with increasing hillslope were more pronounced than the increases in the amount of sediment yield. This greater increase in soil loss than sediment yield, as hillslope increases, has also been found in other studies (El Kateb et al. 2013). As shown in this study, and demonstrated also by Liu et al. (2001), the amount of soil loss that occurs is directly linked to the gradient of the hillslope. Similar to soil loss, sediment yield increased as hillslope increased. Increases to hillslope percent are also expected to increase the amount of sediment yield that occurs (Defersha and Melesse 2012), which was observed in the scenarios simulated.

The simulation results showed that changes in surface runoff, soil loss, and sediment yield are amplified by the combined effects of disturbance strategies and increases to hillslope gradients, especially with the case of wildfire (Fig. 6-8). It appears that the effect of winter grazed land surface disturbance scenarios and increases to hillslope gradient are not as strongly amplified as wildfire land surface disturbance scenarios combined with increases to hillslope gradients.

Table 1: Input rangeland plant parameters used for the baseline scenario which is the conventional summer-long grazing on 2% hillslope. Rangeland plant parameters were held constant for all scenario simulations.

Rangeland Plant Parameters Used for Model Set Up		
Parameter	Value	Units
Change in surface residue mass coefficient	2.09	-
Coefficient for leaf area index	10	-
Change in root biomass coefficient	1.4	-
Parameter for canopy height	4.8	-
Daily removal of surface by insects	0	kg/ha
Fraction of 1st peak to growing season	0.6	-
Fraction of 2nd peak of growing season	0.4	-
c:n ratio of residue and roots	35	-
Standing biomass where canopy cover is 100%	4483	kg/ha
Frost-free period	227	days
Projected plant area coefficient for grasses	0.43	-
Average canopy diameter for grasses	9	cm
Average height for grasses	24	cm
Average number of grasses along a 100m belt transect	700	-
Minimum temperature to initiate growth	5	^oC
Maximum herbaceous plant height	49	cm
Maximum standing live biomass	2242	kg/ha
Plant drought tolerance factor	0.2	-
Day of peak standing crop, 1st peak	152	Julian day
Minimum of live biomass	448	kg/ha
Root biomass in top 10cm	6379	kg/ha
Fraction of live and dead roots from maximum at start of year	0.25	-
Day on which peak occurs, 2nd growing season	213	Julian day
Projected plant area coefficient for shrubs	0	-
Average canopy diameter for shrubs	0	feet
Average height of shrubs	0	feet
Average number of shrubs along a 100m belt transect	0	-
Projected plant area coefficient for trees	0	-
Average canopy diameter for trees	0	feet
Minimum temperature to initiate senescence	-1	^oC
Average height for trees	0	cm
Average number of trees along a 100m belt transect	0	-
Fraction of initial standing woody biomass	0	-

Table 2: Initial field conditions that existed on January 1st of 2017 and represent in the baseline scenario which is the conventional summer-long grazing on 2% hillslope. Initial field conditions were left constant through all scenario simulations.

Initial Field Conditions for Baseline Scenario		
Parameter	Value	Units
Initial frost depth	15	cm
Average rainfall during growing season	24.4	cm
Initial residue mass above the ground	160	kg/ha
Initial residue mass on the ground	100	kg/ha
Initial random roughness for rangeland	3.3	cm
Initial snow depth	7.6	cm
Initial depth of thaw	15	cm
Depth of secondary tillage layer	0	cm
Depth of primary tillage layer	0	cm
Interrill litter surface cover (0-100)	45	%
Interrill rock surface cover (0-100)	0	%
Interrill basal surface cover (0-100)	3	%
Interrill cryptogamic surface cover (0-100)	0	%
Rill litter surface cover (0-100)	45	%
Rill rock surface cover (0-100)	0	%
Rill basal surface cover (0-100)	0	%
Rill cryptogamic surface cover (0-100)	0	%
Total foliar (canopy) cover (0-100)	95	%

Table 3: Soil properties used in the WEPP model for scenario simulations. Initial parameter estimates are the default imported values from the web soil survey (https://websoilsurvey.sc.egov.usda.gov/App/WebSoilSurvey.aspx). Soil parameter values used in the baseline model are adjusted values obtained from calibration of the baseline model.

No.	Parameter	Initial Value	Calibrated value	Units
1	Albedo	23	5%	%
2	Initial Saturation level	75	75%	%
3	Interrill erodibility	4.315	4.315	Kg*s/m**4
4	Rill Erodibility	0.002408	0.0024	s/m
5	Critical Shear	0.0731	0.07311	Pa
6	Effective Hydrologic Connectivity	0.05591	0.05591	mm/hr
	Soil Layer 1			
7	Depth	10.2	25.4	mm
8	Sand %	6.4	6.4	%
9	Clay %	24.5	25.5	%
10	Organic %	1	1	%
11	CEC	16.5	16	meq/100g
12	Rock %	0	0	%
	Soil Layer 2			
13	Depth	152.4	157.5	mm
14	Sand %	35.8	15	%
15	Clay %	27.5	27.5	%
16	Organic %	0.33	0.33	%
17	CEC	16.5	16	meq/100g
18	Rock %	0	0	%

Table 4: Parameter values used for the baseline scenario, which represents conventional summer-long grazing on 2% hillslope.

5/15/2017 - Start Grazing	
Parameter	Value
Pasture Area (ha)	65.5
Maximum Digest of Forage	0.5
Minimum Digest of Forage	0.2
Fraction of Forage Available	0.8
Number of Animals	22
Body Weight (kg)	317
7/13/2017- Stop Grazing	
5/15/2018 - Start Grazing	
Parameter	Value
Pasture Area (ha)	65.5
Maximum Digest of Forage	0.5
Minimum Digest of Forage	0.2
Fraction of Forage Available	0.8
Number of Animals	19
Body Weight (kg)	317
8/15/2018 - Stop Grazing	

Table 5: Parameter values used for the winter grazing scenario for all hillslopes.

1/1/2017 - Start Grazing	
Parameter	Value
Pasture Area (ha)	65.5
Maximum Digest of Forage	0.5
Minimum Digest of Forage	0.2
Fraction of Forage Available	0.8
Number of Animals	150
Body Weight (kg)	317
2/1/2017 - Stop Grazing	
5/15/2017 - Start Grazing	
Parameter	Value
Pasture Area (ha)	65.5
Maximum Digest of Forage	0.5
Minimum Digest of Forage	0.2
Fraction of Forage Available	0.8
Number of Animals	22
Body Weight (kg)	317
7/13/2017- Stop Grazing	
5/15/2018 - Start Grazing	
Parameter	Value
Pasture Area (ha)	65.5
Maximum Digest of Forage	0.5
Minimum Digest of Forage	0.2
Fraction of Forage Available	0.8
Number of Animals	19
Body Weight (kg)	317
8/15/2018 - Stop Grazing	

Table 6: Parameter values used for the wildfire scenario for all hillslopes.

10/15/2017 - Burn	
Parameter	Value
Live biomass Fraction accessible for consumption following burning	0
Fraction reduction in standing wood mass due to burning	1
Fraction change in potential above-ground biomass	0
Fraction evergreen biomass remaining after burning	0
Fraction non-evergreen biomass remaining after burning	0

Table 7: Scenarios simulated to evaluate summer-long grazing, winter grazing, and wildfire events

Scenario	Hillslope	Land Surface Disturbance Scenario	Description
Scenario 1*	2%	CG	Continuous summer-long grazing on 2% slope. This scenario is the baseline scenario and serves as a benchmark to evaluate all other land surface disturbance scenarios.
Scenario 2	2%	WG	Winter-grazing on 2% slope. This scenario evaluates the level of disturbance impact if winter-grazing occurred at a high intensity.
Scenario 3	2%	WF	Wildfire on 2% slope. This scenario evaluates the level of disturbance impact if wildfire were to occur.
Scenario 4	3%	CG	Continuous summer-long grazing on 3% hillslope. This scenario evaluates the level of disturbance impact if the hillslope were to increase from 2% to 3%.
Scenario 5	3%	CG	Continuous summer-long grazing on 5% slope. This scenario evaluates the level of disturbance impact if the hillslope were to increase from 2% to 5%.
Scenario 6	3%	WG	Winter grazing on 3% slope. This scenario evaluates the level of disturbance impact if winter grazing occurs and the hillslope increases from 2% to 3%.
Scenario 7	3%	WF	Wildfire on 3% slope. This scenario evaluates the level of disturbance impact if wildfire were to occur and hillslope were to increase to 3%.
Scenario 8	5%	WG	Winter grazing on 5% slope. This scenario evaluates the level of disturbance impact if winter grazing occurred and the hillslope increases from 2% to 5%.
Scenario 9	5%	WF	Wildfire on 5% slope. This scenario evaluates the level of disturbance impact if wildfire were to occur and hillslope were to increase to 5%.

* Denotes baseline scenario on which the model is calibrated and validated
Calibration Period: January 1st, 2018 - December 31st, 2018
Validation Period: April 2017 13th - December 31st, 2017

Table 8: WEPP model performance statistics for simulation of soil moisture for the calibration (January 1st, 2018 – December 31st, 2018), validation (April 13th, 2017 – December 31st, 2017), and the entire study (January 1st, 2017 – December 31st, 2018) periods.

		Daily	
Statistics	Calibration	Validation	Entire study period
R^2	0.70	0.53	0.53
NSE	0.50	0.47	0.49
		Monthly	
R^2	0.78	0.72	0.51
NSE	0.53	0.57	0.51

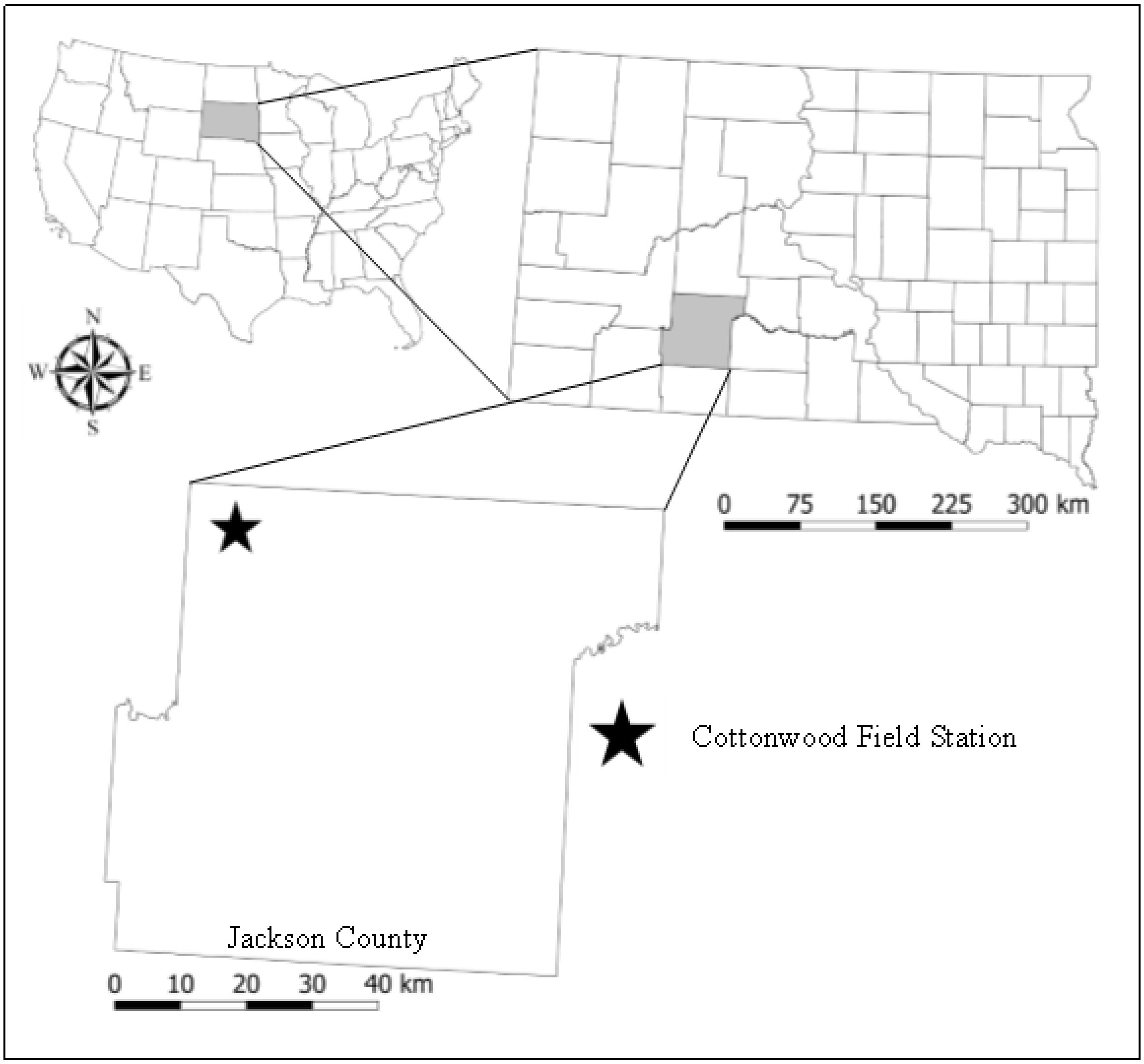


Figure 1:

Location of the Cottonwood Field Station in Cottonwood, South Dakota.

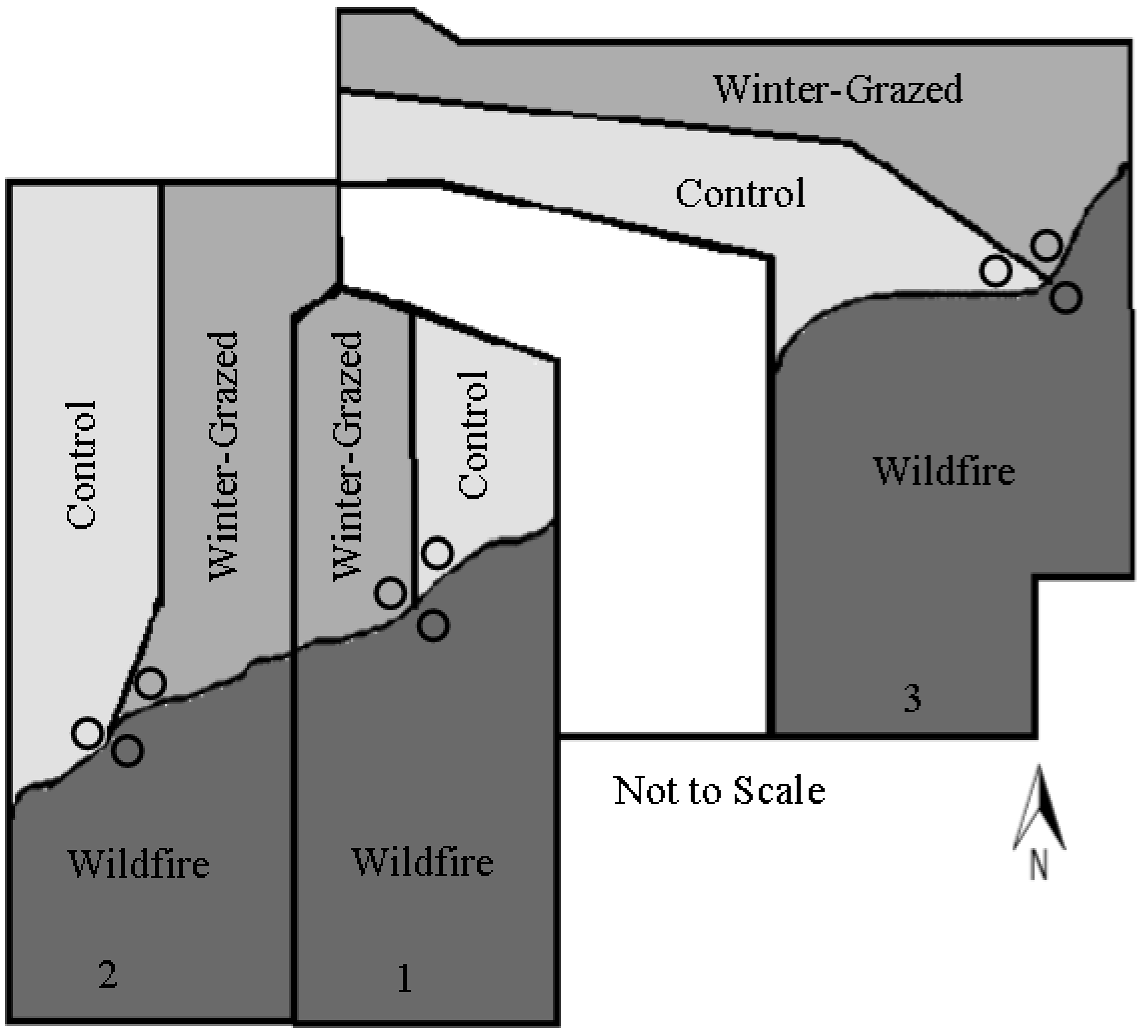


Figure 2:

Map showing field data collection sites at South Dakota State University Agricultural Experiment and Research Station in of Cottonwood, South Dakota to evaluate summer-long continuous gazing (Control), winter grazing, and wildfire disturbances on soil hydrology. Circles indicate the locations of infiltration tests and soil moisture/temperature probes in each pasture. Soil moisture and temperature were collected at three depths (6-, 12-, and 24- inch).

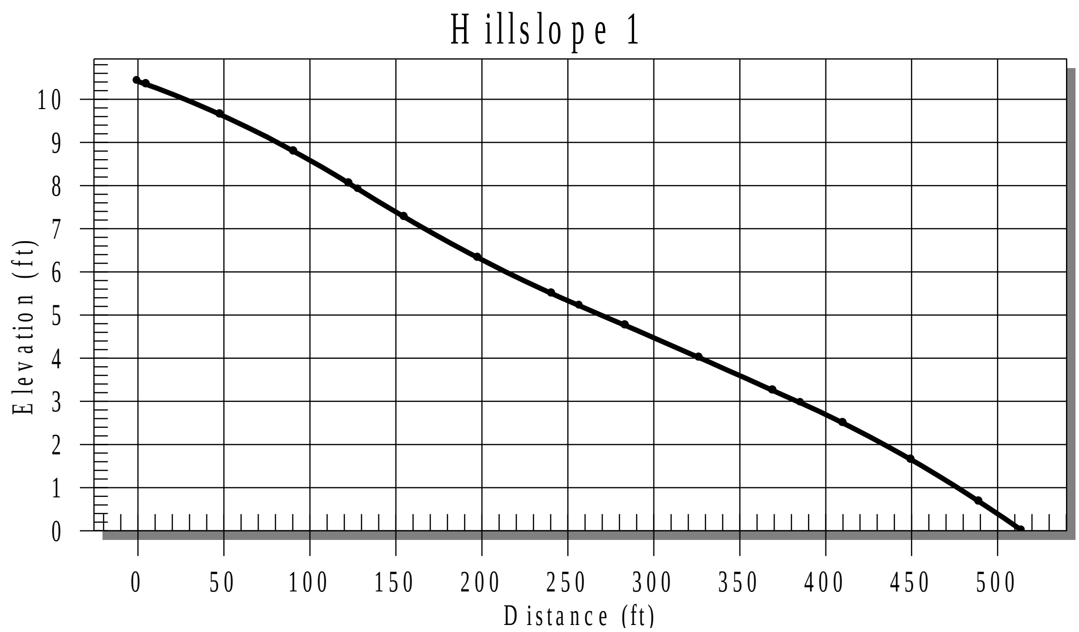


Figure 3:

Hillslope 1 profile (2% slope) as used in the WEPP model simulations.

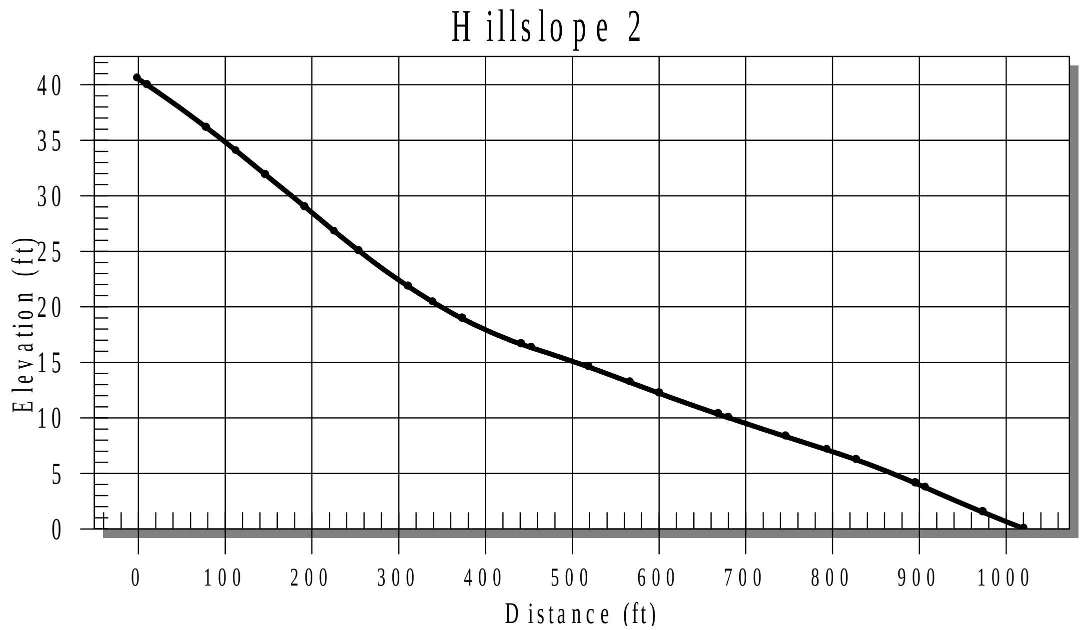


Figure 4:

Hillslope 2 profile (3% slope) as used in the WEPP model simulations.

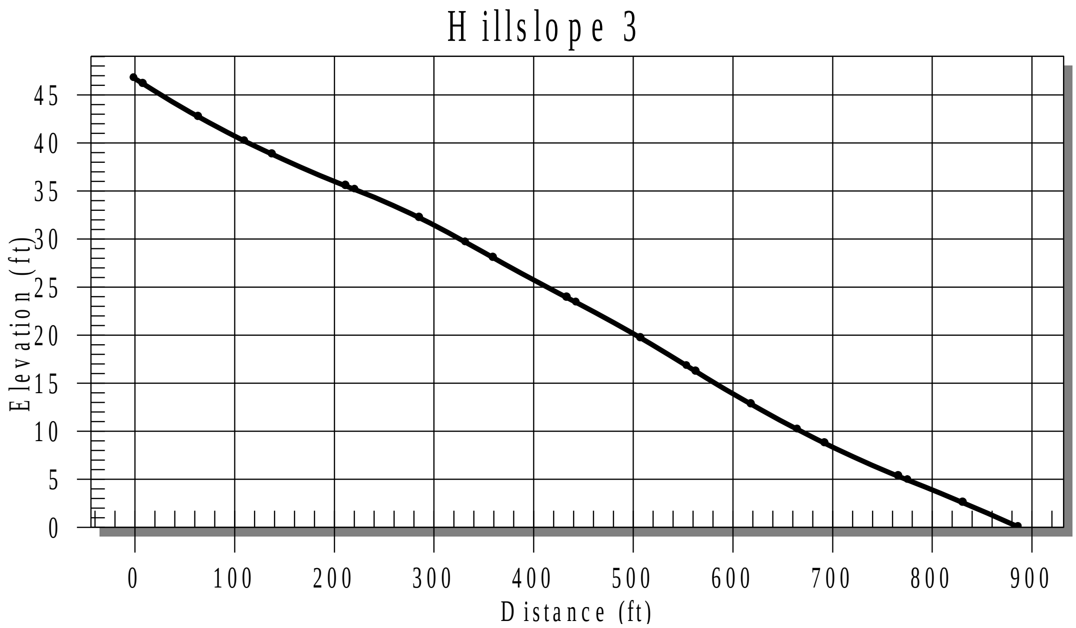


Figure 5:

Hillslope 3 profile (5% slope) as used in the WEPP model simulations.

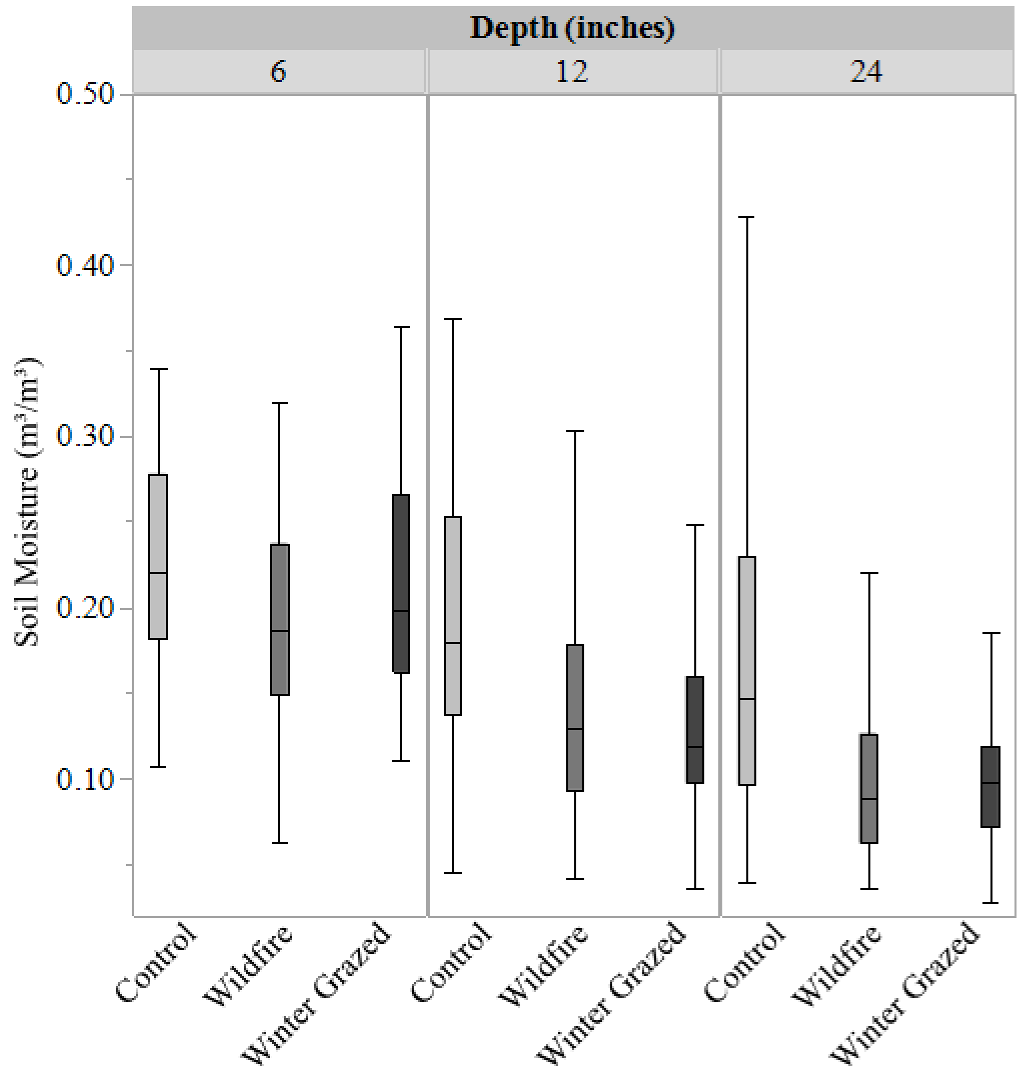


Figure 6:

Median soil moisture (m^3/m^3) for all three land surface disturbance treatments across three depths.

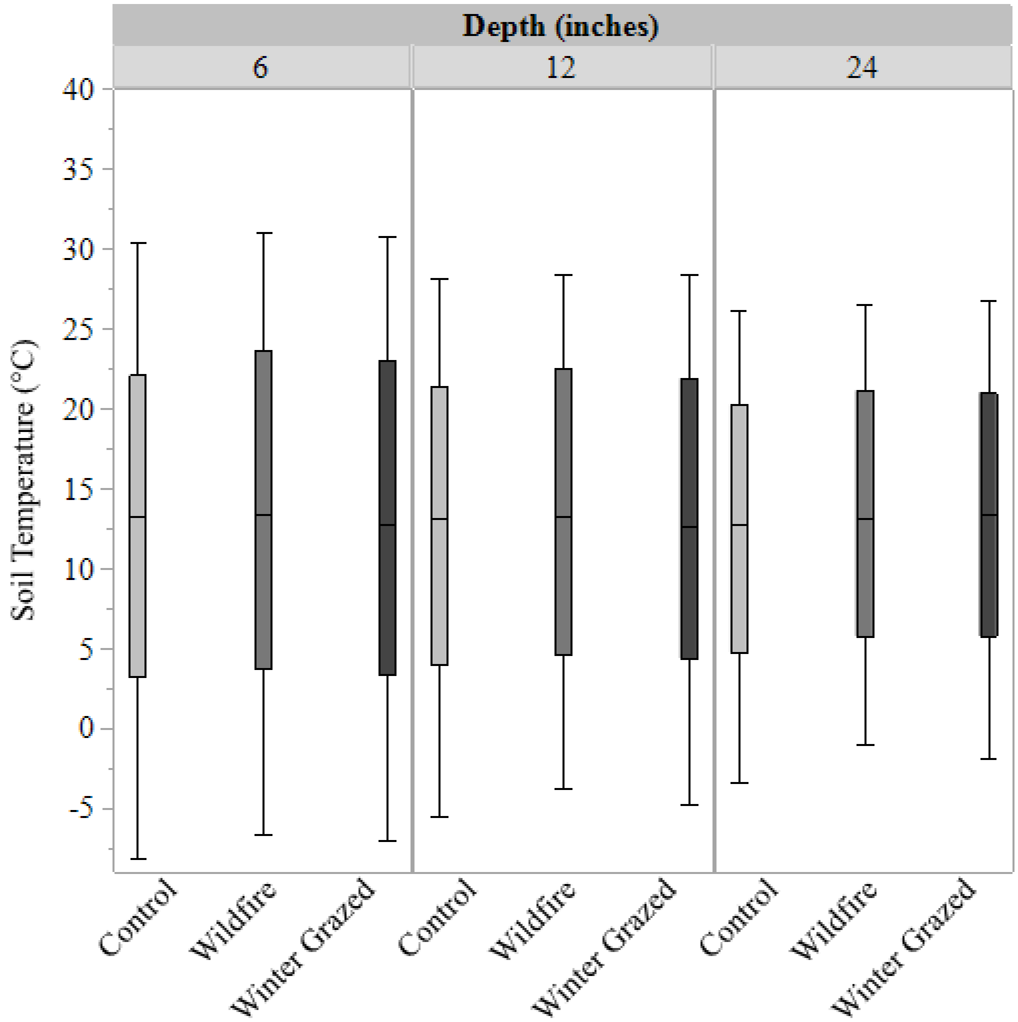


Figure 7:

Median soil temperature (°C) for all three land surface disturbance treatments across three depths.

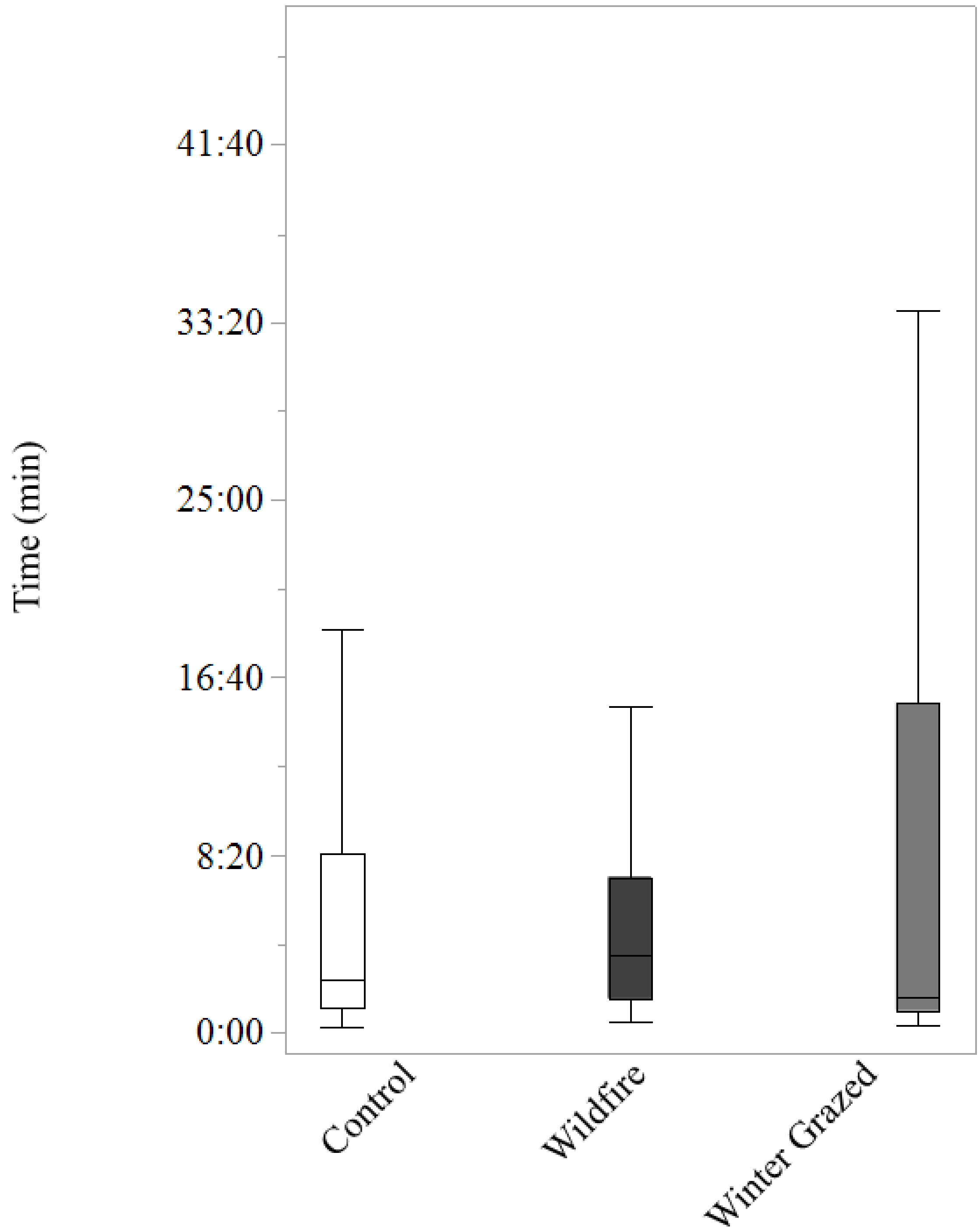


Figure 8:

Median infiltration rates (minutes) for all three land surface disturbance treatments.

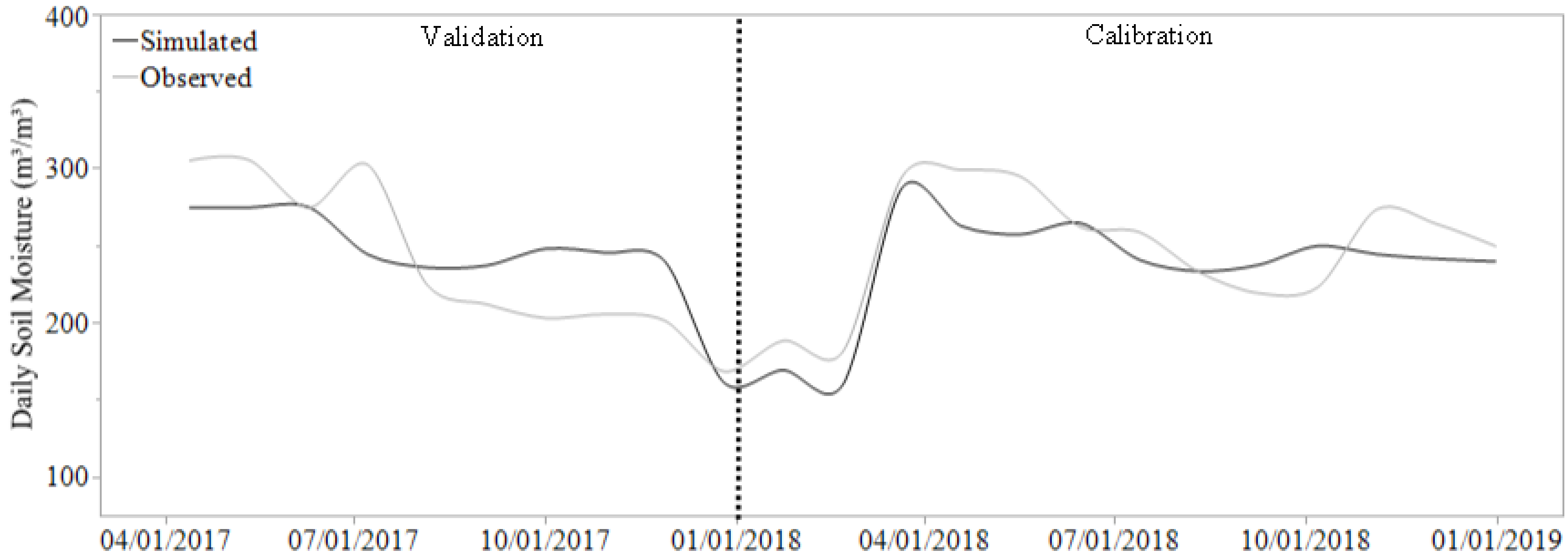
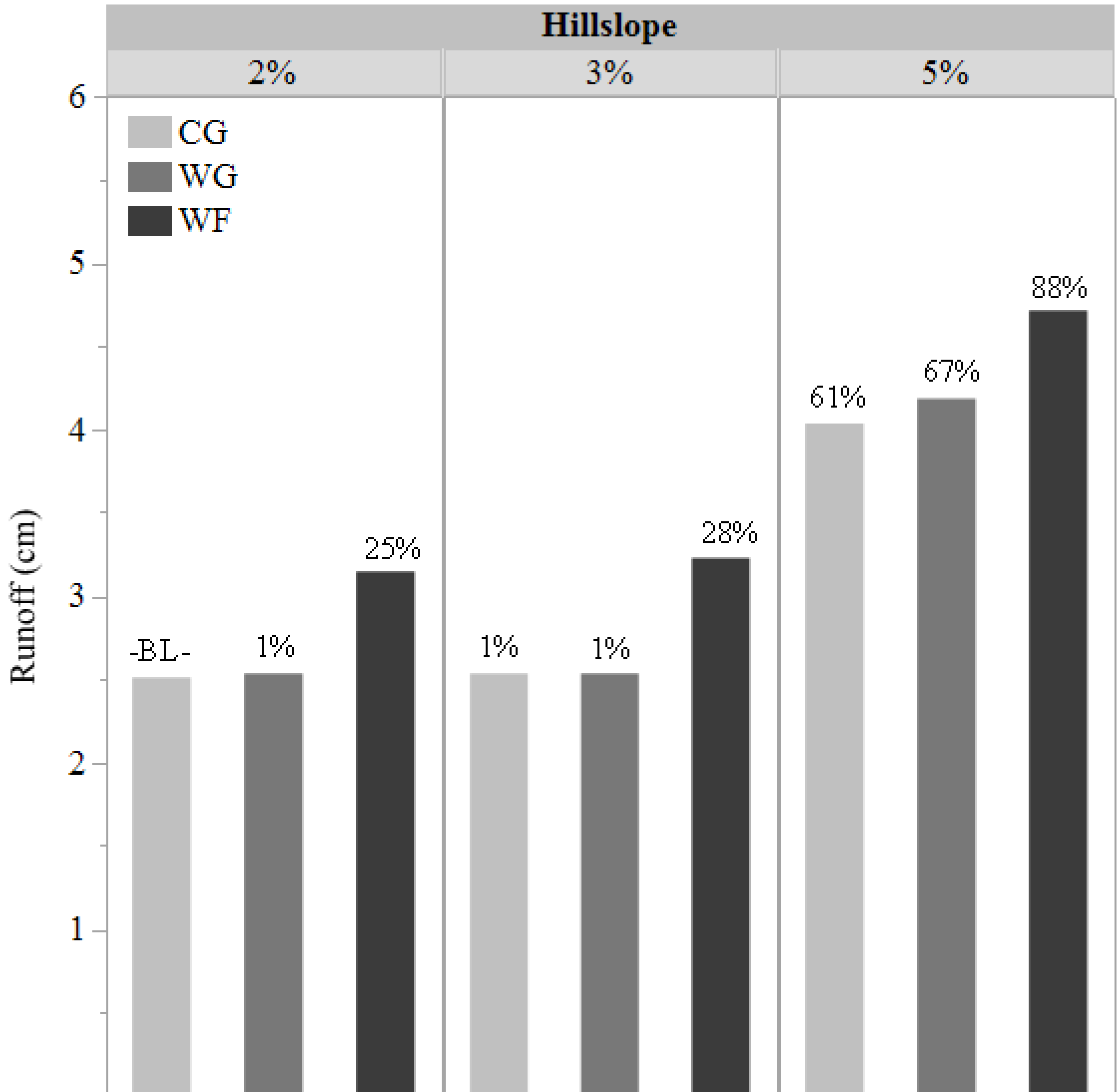


Figure 9:

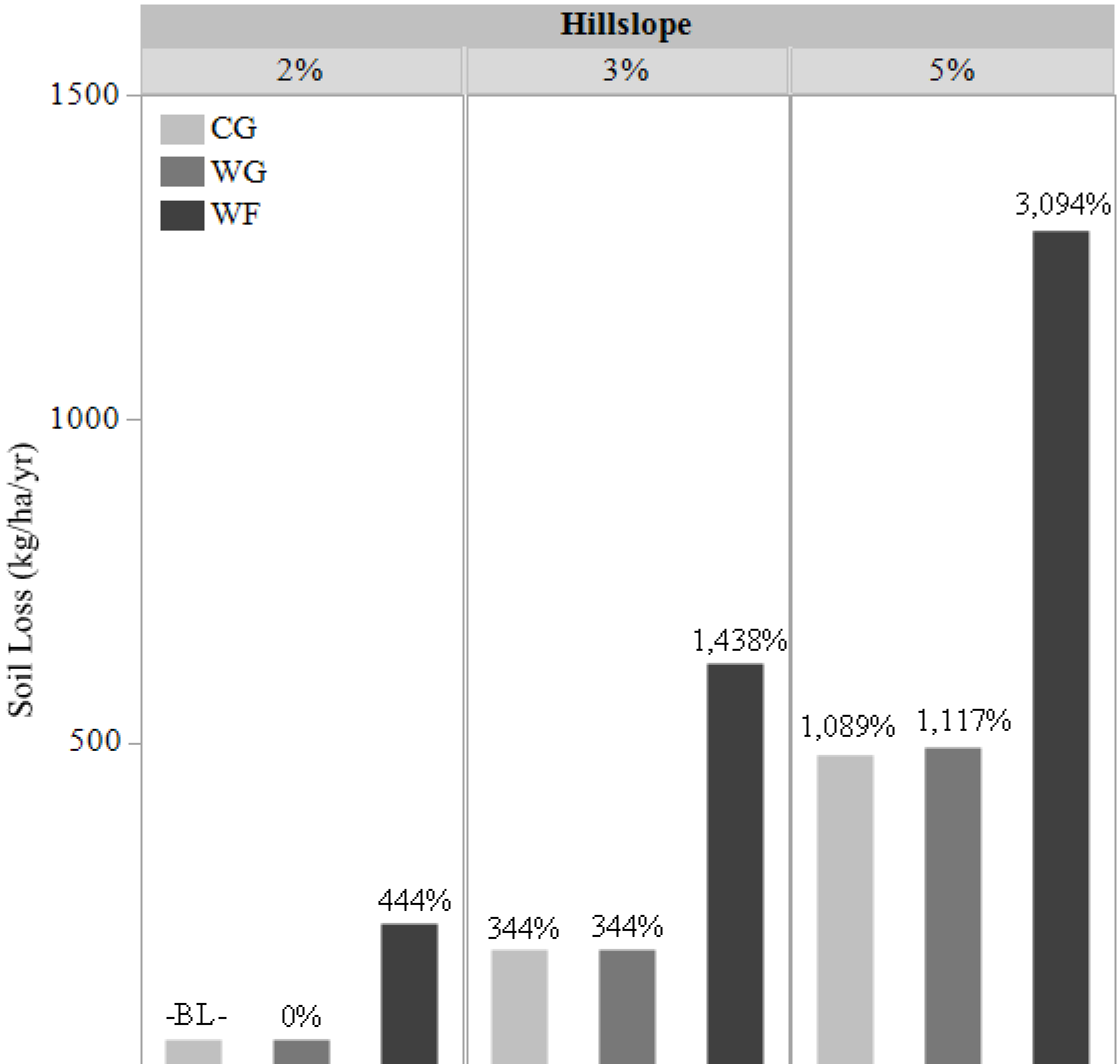
Daily simulated soil moisture compared to observed daily soil moisture for the baseline scenario during the period of April 13th, 2017 through December 31st, 2018.



	2%	3%	5%
CG	2.51	2.54	4.04
WG	2.54	2.54	4.19
WF	3.15	3.23	4.72

Figure 10:

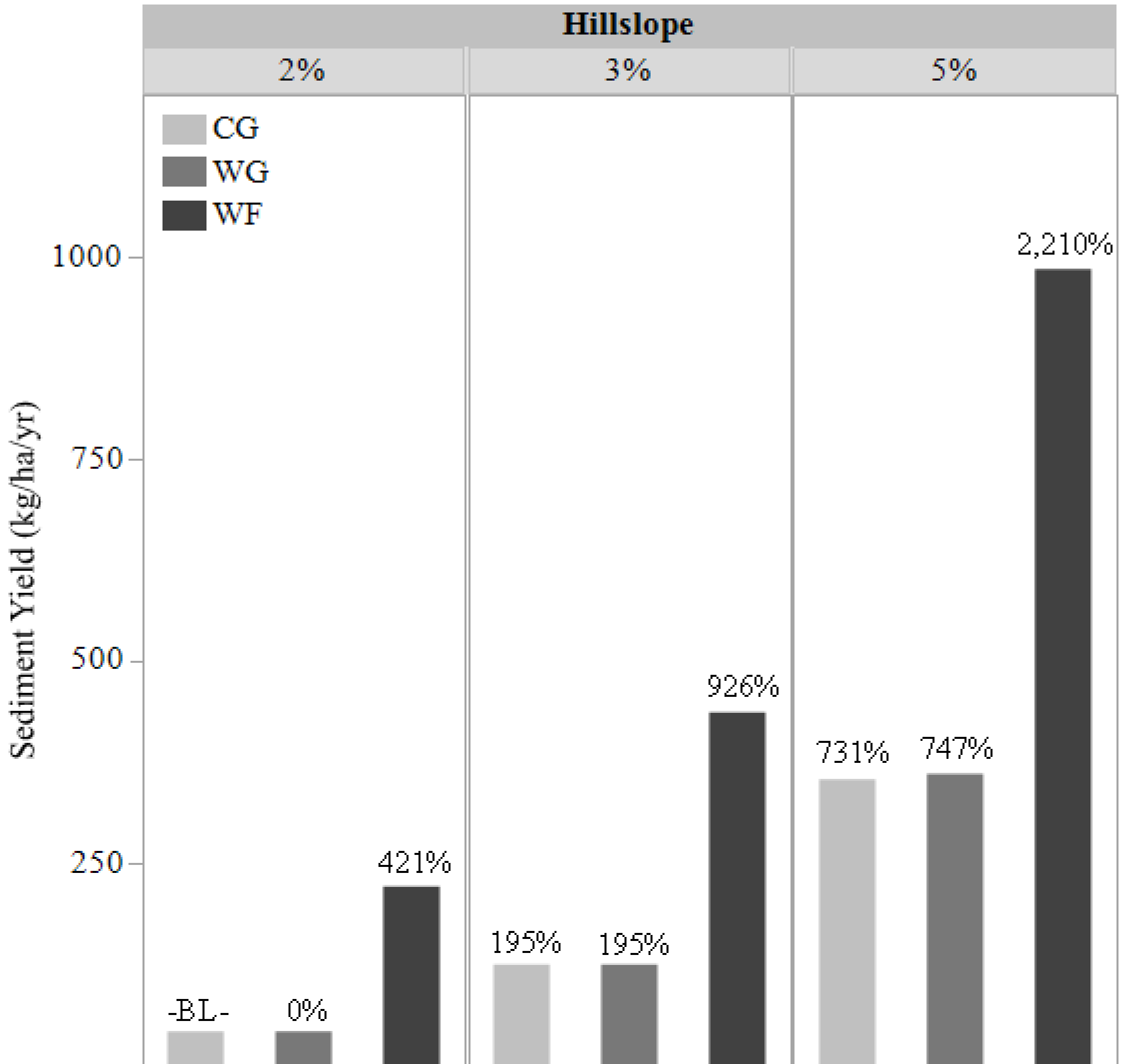
Average annual runoff (mm) of three land surface disturbance scenarios on three hillslopes for the 2017-2018 period. Percent values located above each bar indicate the percent changes of runoff relative to the baseline scenario.



	2%	3%	5%
CG	40.35	179.34	479.72
WG	40.35	179.34	490.93
WF	219.69	620.95	1,288.98

Figure 11:

Average annual soil loss (ton/A/yr) of three land surface disturbance scenarios on three hillslopes for the 2017-2018 period. Percent values located above each bar indicate the percent changes of runoff relative to the baseline scenario.



	2%	3%	5%
CG	42.59	125.54	354.19
WG	42.59	125.54	360.91
WF	221.93	437.13	984.10

Figure 12:

Average sediment yield (ton/A/yr) of three land surface disturbance scenarios on three hillslopes for the 2017-2018 period. Percent values located above each bar indicate the percent changes of runoff relative to the baseline scenario.

CONCLUSION

The overall purpose of my research was to compare the impacts of high-intensity winter grazing, wildfire, and summer-long continuous grazing on soil microbial communities and hydrological processes. My research answers the question: do high-intensity winter grazing or wildfire have detrimental effects on the soil microbial community, soil moisture, soil temperature, infiltration, and erosion processes, which including surface runoff, soil loss, and sediment yield? I found that winter grazing could serve as an alternative grazing strategy that does not cause adverse effects on the soil microbial community, soil moisture, soil temperature, infiltration rates, or erosion processes. Wildfire also did not result in adverse effects to the soil microbial community, soil moisture, and soil temperature, but did negatively impact erosion processes.

Although we found no correlation between surface disturbance treatments and the soil microbial community, the effect of the land surface disturbance treatments did vary depending on vegetation communities. In future studies, the effect high-intensity winter grazing should be evaluated for its effects in different vegetation communities. A closer examination of the soil microbial community, at the species level, may also provide valuable insight into the effects of high-intensity winter grazing and wildfire.

High-intensity winter grazing did not cause significant increases in surface runoff, soil loss, or sediment yield, nor was the impact amplified as hillslope increased. The effect of wildfire on surface runoff, soil loss, and sediment yield was amplified as hillslope increased. With a better understanding of the impacts of high-intensity winter grazing, decisions can be made about the suitability of this alternative grazing strategy to be used in the Northern Great Plains to restore diversity.

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